HANDBOOK OF ENVIRONMENT AND WASTE MANAGEMENT

Land and Groundwater Pollution Control

Volume 2

edited by

Yung-Tse Hung • Lawrence K Wang

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World Scientific

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Also Published by World Scientific

Handbook of Environment and Waste Management
Volume 1: Air and Water Pollution Control
ISBN 978-981-4327-69-5

Handbook of Environment and Waste Management
Volume 2: Land and Groundwater Pollution Control
ISBN 978-981-4449-16-8

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NEW JERSEY • LONDON • SINGAPORE • BEIJING • SHANGHAI • HONG KONG • TAIPEI • CHENNAI

Published by

World Scientific Publishing Co. Pte. Ltd.
5 Toh Tuck Link, Singapore 596224
USA office: 27 Warren Street, Suite 401-402, Hackensack, NJ 07601
UK office: 57 Shelton Street, Covent Garden, London WC2H 9HE

Library of Congress Control Number: 2013956067

British Library Cataloguing-in-Publication Data
A catalogue record for this book is available from the British Library.

HANDBOOK OF ENVIRONMENT AND WASTE MANAGEMENT
Volume 2: Land and Groundwater Pollution Control

ISBN 978-981-4449-16-8

In-house Editors: Dr. Ng Yan Hong/Amanda Yun

Typeset by Stallion Press
Email: enquiries@stallionpress.com

Printed in Singapore

DEDICATION

Professor William Wesley Eckenfelder, Jr., D. Sc., P.E., DEE
(*November 15, 1926–March 28, 2010*)

The editors of the *Handbook of Environment and Waste Management* dedicate this volume to the loving memory of Professor William Wesley Eckenfelder, Jr. D. Sc., P.E., DEE, Distinguished Professor of Environment and Water Resources Engineering, Vanderbilt University, Nashville, Tennessee, the USA. Prof. Eckenfelder passed away on March 28, 2010, in Nashville, Tennessee, the USA. He was 83. He was born in New York City on November 15, 1926, and graduated from high school at age 16. He received bachelors' degree in civil engineering from Manhattan College in 1946. He earned a masters' degree in sanitary engineering from Pennsylvania State University in 1948, and a masters' degree in civil engineering from New York University in 1954. He also pursued post-graduate studies at North Carolina State University and Pennsylvania State University. He was deemed the godfather of industrial wastewater management by his colleagues, former students, and peers. He was an Environmental Engineering Professor at Manhattan College, New York, the USA, the University of Texas at Austin (1965–1969), Texas, the USA, and Vanderbilt

University (1969–1989), Nashville, Tennessee, the USA. He was the best professor and mentor to his students. His office door is always open for his students. He was very caring and helpful to his students. He has touched and changed the lives of his students. He will be missed by all of his students. He was the Ph.D. dissertation supervisor of Prof. Yung-Tse Hung, the editor of *Handbook of Environmental and Management*. Prof. Hung received excellent preparation for his university teaching career from Prof. Eckenfelder. Prof. Hung was Prof. Eckenfelder's last Ph.D. student at the University of Texas at Austin in 1970 and has the same birthday of November 15 as of Prof. Eckenfelder.

PREFACE

The environmental system has existed from the earliest time that life in its primitive forms appeared on this planet of earth. Before the civilization, many animal and plant species emerged, evolved, or become extinct, as environmental system changed. The earth generally purified itself by its unique self-purification process and the availability of natural resources remained unchanged. Civilization has created environmental pollution, especially after the industrial revolution. Air, water, and land in some industrial and developing countries have been heavily polluted to an unacceptable level that Mother Nature can no longer be able to purify itself. As a result, the renewable resources, such as farm lands, rain forests, surface water supplies, groundwater supplies, ocean/lake fisheries, and watersheds, are contaminated by the human activities rapidly. The nonrenewable resources, such as coal, oil, natural gases, metallic ores, and rare nonmetallic ores, are consumed or wasted at an ever-increasing rate and will be exhausted in a few decades, if proper conservation actions are not taken in a timely manner. Radioactive pollution is extremely serious because the normally renewable resources, such as land and groundwater, could become nonrenewable and be almost forever gone, if contaminated by high-level radioactive wastes. Oil and hazardous substances spills on land or in ocean may endanger the ecosystem for a very long time. Destruction of ozone layer by the chlorinated hydrocarbons will increase the dangerous UV exposure. Burning fossil fuels at the current rate will cause global warming and climate changes, in turn, causes the chain reactions of ice melting, land flooding, desert formation, hurricanes, tornadoes, species extinction, ocean current diversion, and perhaps even arrival of another ice age.

Once upon a time, fresh air, palatable water, and beautiful clean land were taken by people for granted worldwide. In many heavily polluted regions now, drinking bottle water instead of tap water has become routine. It would be horrible if one day the human beings would face the situations that (a) the air is contaminated by toxic substances, so we must breath air from the pressurized cylinders; (b) the ozone layer in the sky is destroyed, so we must all wear the sun glasses and special clothing for protection of eye sight and skin, respectively, from the excessive UV lights; (c) the surface and ground water resources are contaminated by acid rain, toxic organics, and heavy metals, so we lose potable water supplies, fisheries, irrigation values, or recreation values; (d) the ocean is polluted by oil spills and ocean waste disposal, so we lose ocean fisheries, aquatic species, beautiful coastal areas, etc.; (e) the land and groundwater are polluted by hazardous substances and solid wastes, so the contaminated sites are no longer inhabitable; and (f) the continuous release of green house gases to the air to cause global warming and climate changes, so we lose lands, many animal and plant species, and may even lose human species if the ice age arrives.

The two volumes of the *Handbook of Environment and Waste Management* series have been developed to deal with the aforementioned environmental pollution problems and to provide proper treatment and waste management solutions. Specifically, the entire handbook series is a comprehensive compilation of topics that are at the forefront of many of the technical advances and practice in controlling pollution in air, surface water, groundwater, and land. The text covers biological, physical, chemical, agricultural, meteorological, medical, radioactive, and legal aspects of environmental engineering. Each volume covers basic and advanced principles and applications and includes figures, tables, examples, and case histories.

Internationally recognized authorities in the field of environment and waste management contribute chapters in their own areas of expertise. The authors who were invited to contribute to this handbook series include the environmental experts from the USA, China, Malaysia, Jordan, Iran, Nigeria, Turkey, Brazil, India, Spain, Cuba, Singapore, Ukraine, France, Australia, Taiwan, Canada, Egypt, Russia, and Poland. The editors believe that the unified interdisciplinary approach presented in the handbook is a logical step in the evolution of environmental pollution control and hope that the handbook series becomes a one-stop reference source for readers to get all necessary technical information on air, water, and land resource managements.

This particular book, Volume 2, *Land and Groundwater Pollution Control*, deals with mainly with control technologies and methods for management of land and groundwater resources and is a sister book to Volume 1, *Air and Water Pollution Control*. This book (Volume 2) covers the subjects of biosolids management, sludge management, solid waste disposal, landfill liners, beneficial reuse of waste products, recycling of foundry sand as construction materials, stabilization of brown coal fly ash using geopolymers, municipal solid waste recovery, reuse of solid wastes as construction materials, biological methods for toxicity evaluation of wastes and waste-amended soils, groundwater contamination at landfill site, remediation of contaminated groundwater, radioactive pollution and control, plastics waste management, and water utility sludge management.

The sister book, Volume 1, *Air and Water Pollution Control*, deals mainly with control technologies and methods for management of air and surface water resources. The sister book introduces the subjects of air pollution and its control, air quality modeling and prediction, air biofiltration for odor treatment, drinking-water-associated pathology, wastewater disinfection, chemical and photochemical advanced oxidation processes, membrane separation for water and wastewater treatment, municipal wastewater treatment and reuse, agricultural irrigation, combine sewer overflow treatment, storm water management, biological wastewater treatment, aerobic granulation process, sequencing batch reactors, environmental impact assessment on aquatic pollution, decentralized sewage treatment technologies, wetland waste treatment technologies, land waste treatment technologies, landfill leachate treatment and management, river and lake pollution control, dye wastewater treatment, olive oil manufacturing waste treatment, medical waste management, environmental enzyme technology, various microorganisms for environmental biotechnology processes, and flotation technologies.

The editors are pleased to acknowledge the encouragement and support received from their colleagues and the publisher during the conceptual stages of this endeavor. We wish to thank the contributing authors for their time and effort and for having patiently borne our reviews and numerous queries and comments. We are very grateful to our respective families for their patience and understanding during some rather trying times. The editors are especially indebted to Mrs. Kathleen Hung Li, who is the daughter of Chief Editor Yung-Tse Hung, and was a manager of the Texas Hospital Association, Austin, Texas, for her services as consulting editor of this handbook series.

Yung-Tse Hung, Ohio, the USA
Lawrence K. Wang, New York, the USA
Nazih K. Shammas, California, the USA
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Chapter 1

BIOSOLIDS MANAGEMENT

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Abstract

Managing biosolids and wastewater is not an easy task. Requirements for higher degrees of wastewater treatment can increase the total volume of biosolids generated. The biosolids management options would be much complicated when the combination of more biosolids quantities, mixtures of biosolids, and increasing regulatory requirements have to be considered. In most of the treatment facilities, a large portion of total operating and maintenance costs is allocated for biosolids processing and disposal. This chapter will discuss in depth about biosolids management starting from the generation of the biosolids until they are ready to be reused or disposed of. The technological topics covered are biosolids production, classification of biosolids (primary, chemical, biological, other wastewater biosolids, etc.), biosolids treatment and processing (thickening, stabilization, conditioning, and dewatering), land application, and finally the use and disposal of biosolids.

Keywords: Biosolids, classification of biosolids, biosolids management, biosolids treatment and processing, land application, use and disposal.

1. Introduction

1.1. *Biosolids*

Biosolids, which is also referred as sewage sludge, can be defined as the residual solids generated from the processing of domestic wastewater that meet the regulatory requirements for recycling.[1] In most cases, this product can be used beneficially. Generally, domestic wastewater consists of wastes in the form of liquid produced from residences, businesses, and institutions. Any of the materials that enter the municipal wastewater collection system finally end up into biosolids. It is known that biosolids are rich in nutrients and are good source of natural fertilizer to stimulate plant growth or soil amendment to enhance the land. Normally, biosolids contain significant quantities of water, organic matter, nutrients, and trace elements. According to U.S Environmental Protection Agency, biosolids contain about 93–99% of water, solids, and dissolved matters present in the wastewater or added during wastewater or biosolids treatment processes.[2] Meanwhile, animal manures, untreated septage, municipal solid waste (MSW), untreated wastewater sludges, hazardous wastes, industrial waste, grit, and screenings removed during the initial wastewater treatment process are the substances that are not included under biosolids. The presence of biosolids in the wastewater treatment facility is influenced by the raw sewage that enters the unit that may be contaminated by the chemicals, microorganisms, and also the heavy metals. If the biosolids produced will be used as a fertilizer, we should know and control the contaminants entering the wastewater treatment plant (WWTP).

1.2. *Biosolids Production*

Biosolids are produced during the wastewater treatment. The wastewater treatment can begin before the wastewater reaches the treatment plant. Generally, before the wastewater is released to the WWTP, the wastewater must be pretreated first in order to remove any hazardous contaminants including metals such as copper, lead, cadmium, and chromium and other pollutants. Over the past 20 years, pretreatment and pollution prevention programs have reduced the level of metals and other pollutants going into WWTP. This can help to improve the quality of biosolids produced.[2]

Once the wastewater reaches the WWTP, it will undergo preliminary, primary, secondary, and tertiary treatment. Table 1 lists the types of wastewater treatment

Table 1. Types of Wastewater Treatment and Types of Biosolids Produced.[2]

Wastewater Treatment Level	Types of Biosolids Produced
Screening and grit removal (preliminary treatment)	
Wastewater screening removes coarse solids that can interfere with mechanical equipment. Grit removal separates heavy, inorganic, and sand-like solids that would settle in channels and interfere with treatment processes.	Screenings and grit are handled as a solid waste and nearly always landfilled.
Primary wastewater treatment	
Usually involves gravity sedimentation of screened and degritted wastewater to remove SS before secondary treatment.	Biosolids produced at this stage usually contain 3–7% solids. Normally, water content can be easily reduced by thickening or dewatering.
Secondary wastewater treatment	
Involves biological treatment process such as suspended growth or fixed growth system. During the biological treatment, microorganisms are used to reduce BOD and remove SS.	Biosolids produced usually have a low solids content (0.5–2%). The products are more difficult to thicken and dewater compared to primary biosolids.
Tertiary wastewater treatment	
Common types of tertiary treatment include biological and chemical precipitations. The processes are used to remove nitrogen and phosphorus.	Lime, polymers, iron, or aluminum salts used in tertiary treatment produce biosolids with varying water-absorbing characteristics.

and the types of biosolids produced after each treatment. Basically, the quantity and characteristics of the biosolids produced at the WWTPs depend on three important factors that include the composition of wastewater, the type of wastewater treatment used, and the type of treatment applied to the biosolids. Generally, the total volumes of biosolids generated are influenced by the degrees of wastewater treatment. The higher the level of treatment is, the higher will be the concentrations of contaminants produced in the biosolids. This is because most of the components removed from the wastewater finally end up in the biosolids. Besides that, the addition of chemicals to precipitate solids for example ferric chloride, lime, or polymers can increase the concentrations of these chemicals in the biosolids produced at the end of the process. Indirect effects also can occur such as when alum (aluminum hydroxide) adsorbs trace metals such as cadmium to precipitate out of the wastewater and into the biosolids. Thus, the type of wastewater treatment or pretreatment used will affect the characteristics of biosolids and also can affect the types of biosolids treatment chosen.

1.3. *Why Biosolids Need to be Treated?*

The biosolids produced can harm and damage when they are released to the environment without proper treatment. In order to treat them, we must have a system to treat the volume of material removed. In addition, releasing the wastewater solids without any treatment would prevent the purpose of environmental protection from being achieved. There are a few treatment systems that can be used to treat the wastewater solids. The purpose of treatment will be the same where to convert the wastewater solids into a form that can be disposed of without giving any harm to the environment or creating inconvenience conditions. In this case, it does not matter what the system or combination of systems chosen for the treatment.

Of the constituents removed during wastewater treatment processes, biosolids are the largest by volume, creating a complex problem for their treatment and disposal. Typically, biosolids contains organic matter (protein, carbohydrates, fats, oils, greases, chemicals, etc.), pathogens and microorganisms (bacteria, viruses, parasites, etc.), heavy and toxic metals, and toxins that include pesticides, household, and industrial chemicals.[3] All of these will give a risk and hazards to humans and the environment. In order to overcome these problems, it is important to treat the biosolids in a proper manner. The followings are the objectives of biosolids treatment:

- reduction or stabilization of organic matter,
- reduction in volume and weight,
- destruction of pathogenic microorganisms and bacteria,
- removal of toxic elements (heavy metals),

- removal of odor, and
- preparation of biosolids for further utilization or disposal.

1.4. *The Importance of Biosolids Management*

Generally, managing biosolids and wastewater is not an easy task. In addition, biosolids management also involves a lot of money to be spent. The greater volumes of biosolids will be produced as higher degrees of wastewater treatment have been required. The biosolids management options would be much complicated when the combination of more biosolids quantities, mixtures of biosolids, and increasing regulatory requirements have to be considered. In most of the treatment facilities, a large portion of total operating and maintenance costs is spent for biosolids processing and disposal.

Biosolids generation and management methods generally depend on the following four factors[3]:

i. sources, quantities, and characteristics of the wastewater,
ii. treatment processes (thickening, stabilization, conditioning, and dewatering),
iii. regulatory public health and environmental considerations, and
iv. performance and costs.

In addition, general considerations and factors in overall biosolids management are listed as below[3]:

- Production and properties of biosolids
  The production and properties of biosolids are mainly contributed by operations such as primary, secondary, and chemical treatments or as well as grit and scum removal and screenings
- High-temperature processes
  These include flash drying, rotary kilns, incineration, starved air combustion, and cogeneration.
- Thickening
  This process consists of gravity thickening, floatation thickening, centrifugal thickening, gravity belt thickening, and rotary drum thickening. All processes play a major role in sedimentation, clarification, flotation, and centrifugation the biosolids.
- Drying
- Stabilization
  This includes the processes such as alkaline stabilization, anaerobic digestion (AD), aerobic digestion, composting, and heat drying.
- Composting
  This process consists of windrow, aerated static piles, and mechanical/in-vessel methods

- Disinfection
  This process includes thermal, chemical, and irradiation processes.
- Conditioning
  This process involves inorganic, organic, elutriation, and thermal techniques.
- Dewatering
  This process includes drying beds, lagoons, and centrifuges.
- Utilization
  This includes cropland, land reclamation, fuel, and raw material recovery.
- Disposal
  This process consists of landfilling, dedicated land disposal (DLD), and ocean disposal.

2. Classification of Biosolids

2.1. *Introduction*

Basically, the generation of wastewater is not only comes from the domestic use, but also comes from the commercial activities, residential areas, and from the industrial sectors. Therefore, the characteristics of wastewater produced will vary from one activity to other. The treatments of wastewater by using various methods such as physical, chemical, and biological processes to eliminate organic and inorganic substances result in the production of biosolids that will vary in quantity and characteristics.

Generally, most biosolids undergo additional treatment on site before they are used or disposed of to meet regulatory requirements that protect public health and the environment, facilitate handling, and reduce costs. Biosolid characteristics can determine a choice of use or disposal methods. To treat and dispose the biosolids that are produced from the WWTP in the most effective manner, it is important to know the characteristics of the biosolids that will be processed. The characteristics of biosolids produced will vary and depends on the origin of the solids and biosolids, the amount of aging that has taken place, and the type of processing that they have been required.[4] Typically, the biosolids produced from the wastewater treatment process can be classified as primary, biological, and chemical biosolids.[5,6] Biosolids usually consist of settleable solids such as fecal materials, silt, fibers, biological flocs, food wastes, organic and inorganic compounds, heavy metals, and trace minerals.

2.2. *Primary Biosolids*

Primary sedimentation has been used widely throughout the world in most WWTPs to remove readily settleable solids from raw wastewater. It was found that the dry weight of primary biosolids is about 50% of the total sludge solids that was produced from a treatment plant equipped with primary sedimentation and a conventional activated sludge process for secondary treatment. Usually primary biosolids are easier to manage compared to biological and chemical biosolids and the reasons are as follows:

- Primary biosolids are readily thickened by gravity.
- Primary biosolids with low conditioning requirements can be mechanically dewatered rapidly.
- The dewatering device will produce a drier cake and it will give better solids capture than the biological and chemical biosolids.

2.2.1. *Primary Biosolids Production*

Typically, primary biosolids production is within the range of 100–300 mg/L of wastewater. There are two basic approaches to estimate primary biosolids production that include: (1) computing the quantity of total suspended solids (TSS) entering the primary sedimentation tank and (2) assuming an efficiency of removal. Usually, estimates of 0.07–0.11 kg/capita/day of TSS are commonly used when site-specific data are not available.[5] Meanwhile, the removal efficiency of TSS in the primary sedimentation tank is usually in the range of 50–65%.[4] For estimating purposes, the removal efficiency of 60% is commonly used. This is subjected to the following conditions:

- The biosolids are mainly produced from a domestic wastewater treatment without major industrial loads.
- The biosolids did not contain chemical from the coagulation and flocculation process.
- No other biosolids have been added to the influent wastewater such as trickling filter biosolids.
- The biosolids did not contain major sidestreams from biosolids processing; for example, digester supernatant, elutriate, and filtrates or centrates and other biosolids such as waste-activated sludge (WAS).

2.2.2. *Factors Affecting the Removal of Solids*

There are several factors that can affect the removal of solids in primary sedimentation, which include:

2.2.2.1. Industrial waste effect

The efficiency of suspended solids (SS) removal in primary sedimentation is depends on the nature of the solids. It is difficult to predict about the effect that industrial

SS can have on removal efficiency. This effect can be seen in the example involved the North Kansas City Wastewater Treatment Plant in Missouri. This plant serves residential areas and numerous major industries such as food processing, paint manufacturing, soft-drink bottling, paper manufacturing, and grain storage and milling. It was found that the raw wastewater that enters the plant had a 15-day average SS concentration of 1,140 mg/L that is produced by the industries. About 90% of these solids have been removed from the primary sedimentation. The quantity of primary sludge was about 1,000 mg/L of wastewater treated while on day 2, the removal exceeded 1,700 mg/L.[7]

2.2.2.2. Ground garbage effect

The use of home garbage grinders can contribute to the increase of the SS load in the WWTP. These solids are largely settleable. It was estimated that the use of home garbage grinders can increase the primary biosolids production in the range of 25% to 50%.[5]

2.2.2.3. Other biosolids and sidestreams

Basically, the amount of biosolids discharged from the primary sedimentation tank is increased when biosolids treatment process sidestreams such as digester supernatant, elutriate and filtrates or centrates, and other biosolids such as WAS are recycled to the primary sedimentation tank. The measurement of solids quantity that entering and leaving the primary clarifier by all streams is an important tool for estimating primary biosolids production when recycled biosolids and biosolids process sidestreams contribute large quantities of solids.

2.2.2.4. Chemical precipitation and coagulation

During the coagulation process, chemicals such as aluminium sulfate or ferric chloride are usually added into the raw wastewater to remove phosphorus or to coagulate the SS. This will result in the formation of a large quantity of chemical precipitates. The quantity of chemical precipitates produced during this treatment depends on several factors such as the type and amount of chemical added, chemical constituents in the wastewater, and performance of the coagulation and clarification processes. However, it is difficult to predict accurately the quantity of chemical solids that will be produced. In most of wastewater laboratory, jar test is found to be the most suitable test to estimate the chemical biosolids quantities. Table 2 lists the quantities of SS and chemical solids removed in a primary sedimentation tank. During this treatment, the chemicals such as lime, aluminium sulfate, or ferric chloride are added into the raw wastewater.

Table 2. Quantities of Suspended and Chemical Solids Removed in a Primary Sedimentation Tank.[5]

		Chemical Addition[a]		
Solids Type	No Chemical Addition[b]	Lime[c]	Alum[d]	Ferric Chloride[e]
Suspended solids, (lb/MG)	1041	1562	1562	1562
Chemical solids, (lb/MG)	—	2082	362	462
Total sludge production, (lb/MG)	1041	3644	1924	2024
(kg/m^3)	(0.13)	(0.44)	(0.23)	(0.24)

[a] Assume 10 mg/L influent phosphorus concentration (as P) with 80% removed by chemical precipitation.
[b] Assume 50% removal of 250 mg/L influent TSS in primary sedimentation.
[c] 125 mg/L $Ca(OH)_2$ added to raise pH to 9.5.
[d] 154 mg/L $Al_2(SO_4)_3 \cdot 14H_2O$ added.
[e] 84 mg/L $FeCl_3$ added.
Note: Assume no recycle streams (for example, recycle of WAS to primary sedimentation, digester, supernatant, etc.). Secondary solids production would be cut from 833 lb/MG without chemical addition to 312 lb/MG with chemical addition in this plant.

2.2.2.5. Peak loads

Generally, peak rates of primary biosolids production can be several times the average. In addition, peak solids production levels also vary from one plant to other.

2.2.3. *Characteristics of Primary Biosolids*

Basically, most primary biosolids can be concentrated readily within the primary sedimentation tanks. It is found that a concentration of 5–6% solids is accomplished when biosolids are pumped from well-designed primary sedimentation tanks.[4] However, the concentration values that are higher or lower than 5–6% range are common. The conditions that can affect the concentration of primary biosolids are as follows[5]:

- The grit may be removed by passing the raw primary biosolids through cyclonic separators if the wastewater is not degritted before it enters the sedimentation tanks. However, if the biosolids concentrations exceed 1%, these separators do not function properly.
- If the biosolids contain large amounts of fine nonvolatile solids such as silt, a concentration of well over 6% may sometimes be achieved.
- Primary biosolids concentration is strongly affected by industrial loads produced from the industrial activities.
- Under anaerobic conditions, primary biosolids may float or rise to the surface of wastewater when buoyed up by gas bubbles. There are few conditions that will encourage gas formation, which include warm temperatures, solids deposits

within sewers, strong septic wastes, long detention times for wastewater solids in the sedimentation tanks, lack of adequate prechlorination, and recirculating sludge liquors. In order to prevent the septic conditions, it is necessary to limit the storage time of biosolids in the sedimentation tanks. To achieve this, the frequency and rate of primary biosolids pumping should be increased.
- A lower primary biosolids concentration will be produced if biological biosolids are mixed with the wastewater.

Table 3 shows the characteristics of primary biosolids. The characteristics of primary biosolids usually consist of several parameters such as pH, volatile acids, heating value, specific gravity (individual solid particles), bulk specific gravity, BOD_5/VSS ratio, COD/VSS ratio, organic N/VSS ratio, volatile content, cellulose, hemicellulose, lignin, grease and fat, protein, nitrogen, phosphorus, and potash. Other than these parameters, it was found that primary biosolids may also contain with some grit, even though the wastewater has been processed through degritting. In addition, the fragmented screenings appear in the primary biosolids when the screenings are comminuted and returned to the wastewater flow. In addition to grit and screenings, smaller plastic and rubber materials that pass through screens also appear in the primary biosolids. Typically, primary biosolids also contain with over 100 different types of anaerobic and facultative species of bacteria. Besides that, sulfate-reducing and -oxidizing bacteria, worm, fly eggs, and pathogenic microorganisms are also present.

2.3. *Chemical Biosolids*

2.3.1. *Introduction*

In wastewater treatment process especially in industrial wastewater treatment, chemicals such as ferric chloride, alum, lime, or polymer are used widely to precipitate the solids and to improve SS removal. The addition of chemicals can result in the formation of chemical biosolids at the treatment plants. Most plants apply chemicals to secondary effluent and use tertiary clarifiers to remove the chemical precipitates. In certain cases, some treatment plants add the chemicals to a biological process. Thus, chemical precipitates are produced and mixed with the primary biosolids or biological biosolids. Chemicals can greatly influence the increasing of biosolids generation and it depends on the chemicals used and the chemicals addition rates. The followings are several types of precipitates that are produced and must be considered in measuring the total biosolids generation[5,8]:

- Phosphate precipitates
  These type of precipitates include $AlPO_4$ or $Al(H_2PO_4)(OH)_2$ with aluminum salts, $FePO_4$ with iron salts, and $Ca_3(PO_4)_2$ with lime.

Table 3. Primary Biosolids Characteristics.[5]

Characteristics	Range of Values	Typical Value	Comments
pH	5–8	6	—
Volatile acids, mg/L as acetic acid	200–2000	500	—
Heating vale, Btu/lb (kJ/kg)	6800–10000	—	Depends upon volatile content and sludge composition, reported values are on a dry weight basis
		10285	Sludge 74% volatile
		7600	Sludge 65% volatile
Specific gravity of individual solid particles	—	1.4	Increases with increased grit, silt, etc.
Bulk specific gravity (wet)	—	1.02	Increases with sludge thickness and with specific gravity of solids
		1.07	Strong sewage from a system of combined storm and sanitary sewers
BOD_5/VSS ratio	0.5–1.1	—	—
COD/VSS ratio	1.2–1.6	—	—
Organic N/VSS ratio	0.05–0.06	—	—
Volatile content (% by weight of dry solids)	64–93	77	Value obtained with no sludge recycle, good degritting; 42 samples, standard deviation 5
	60–80	65	
	—	40	Low value caused by severe storm inflow
	—	40	Low value caused by industrial waste
Cellulose (% by weight of dry solids)	8–15	10	—
	—	3.8	—
Hemicellulose (% by weight of dry solids)	—	3.2	—
Lignin (% by weight of dry solids)	—	5.8	—
Grease and fat (% by weight of dry solids)	6–30	—	Ether soluble
	7–35	—	Ether extract
Protein (% by weight of dry solids)	20–30	25	—
	22–28	—	—
Nitrogen (% by weight of dry solids)	1.5–4	2.5	Expressed as N
Phosphorus (% by weight of dry solids)	0.8–2.8	1.6	Expressed as P_2O_5. Divide values as P_2O_5 by 2.29 to obtain values as P
Potash (% by weight of dry solids)	0–1	0.4	Expressed as K_2O. Divide values as K_2O by 1.20 to obtain values as K

Note: 1 Btu/lb = 2.32 kJ/kg.

- Carbonate precipitates
  The precipitates produced are significantly related to lime treatment. The addition of lime can result in the formation of calcium carbonate, $CaCO_3$.
- Hydroxide precipitates
  The addition of iron and aluminum salts may result the formation of $Fe(OH)_3$ or $Al(OH)_3$. While, magnesium hydroxide $Mg(OH)_2$ is formed when lime is added in the treatment.
- Inert solids from the chemicals
  These inert solids are most significant with lime. A few chemicals supplied in dry form may contain significant amounts of inert solids. For example, if a quicklime is 92% CaO, the remaining 8% may be inert solids that that will end up in the biosolids.
- Polymer solids
  Polymers are widely used in coagulation process as a primary coagulant. Besides that, they play a significant role in order to improve the performance of other coagulants. The addition of polymers in the wastewater during the treatment process may contribute little to total mass. However, they can greatly improve clarifier efficiency with an increase in biosolids production.
- SS from the wastewater
  Process efficiency is affected by the addition of chemicals to a wastewater treatment process.

Generally, the quantities of the various precipitates in chemical biosolids are determined by conditions such as pH, mixing, reaction time, water composition, flocculation, wastewater flow, and chemical dosage.[5] Besides that, changes in wastewater chemistry may also affect the production of chemical biosolids.

2.3.2. *Characteristics of Chemical Biosolids*

The characteristic of chemical biosolids are mainly affected by the precipitated compounds and also by the other wastewater solids. For example, the dewatering process of a lime primary biosolids are much better and easier than the dewatering process of a lime sludge containing large amounts of WAS solids. Generally, the addition of lime during chemical treatment will produce biosolids that thickens and dewaters better compared to the same biosolids without adding any chemicals. The primary biosolid does not thicken or dewater as well as nonchemical biosolid when iron or aluminum salts are added to the raw wastewater. It was found that iron sludges dewater slightly more easily compared to aluminum sludges.[8] Meanwhile in an activated sludge, the sludge may thicken much better than nonchemical activated sludge when the aluminum salts are added. It was found that the thickening and dewatering properties of chemical biosolids can be improved by using anionic polymers during wastewater treatment.

2.4. ***Biological Biosolids***

2.4.1. *General Characteristics*

Generally, biological biosolids are produced from the treatment processes such as activated sludge, trickling filters, and rotating biological contactors. The metabolic and growth rates of microorganisms will affect the quantities and characteristics of biological biosolids. The quantity and quality of biosolids produced by the biological process are intermediate between that produced in non-primary systems and that produced in full-primary systems in cases when fine screens or primary sedimentation tanks with high overflow rates are used. Biological biosolids containing with grit, plastics, papers, and fibers are produced at treatment plants lacking with primary treatment. Normally, pure biological biosolids are produced at the treatment plants with primary sedimentation unit. The method of operation of the clarifiers will affect the concentrations and the volumes of biological biosolids produced. Typically, biological biosolids are more difficult to thicken and dewater compared to primary biosolids and chemical biosolids.

2.4.2. *Activated Sludge*

The variations of an activated sludge include extended aeration, oxidation ditch, pure oxygen, mechanical aeration, diffused aeration, plug flow, contact stabilization, complete mix, step feed, nitrifying activated sludge, etc.[4]

The quantity of WAS is affected by two parameters, which include the dry weight and the concentration of the biosolids. The dry weight of activated sludge production may be predicted. The followings are the important variables in predicting WAS production that include:

(i) the amounts of organics removed in the process,
(ii) the mass of microorganisms in the system,
(iii) the biologically inert SS in the influent to the biological process, and
(iv) the loss of SS to the effluent.

These variables can be used into two simple and useful equations:

$$P_x = (Y)(s_r) - (k_d)(M) \tag{1}$$

$$WAS_T = P_x + I_{NV} - E_T, \tag{2}$$

where P_x = net growth of biological solids (expressed as VSS), lb/day or kg/day; Y = gross yield coefficient, lb/lb or kg/kg; S_r = substrate (for example, BOD5) removed, lb/day or kg/day; k_d = decay coefficient, day^{-1}; M = system inventory of microbial solids (VSS) microorganisms, lb or kg; WAS_T = WAS production, lb/day or kg/day; I_{NV} = non-VSS fed to the process, lb/day or kg/day; and E_T = effluent SS, lb/day or kg/day.

2.4.2.1. Factors affecting the production of WAS

The production of WAS is mainly affected by few factors such as[5]:

- sludge age and F:M ratio,
- nitrification,
- feed composition,
- dissolved oxygen (DO) concentration,
- temperature, and
- feed pattern.

2.4.2.1.1. *Effect of sludge age and F:M ratio*

Equation (1) can be rearranged to show the effect of the sludge age (θ_m).

$$P_x = \frac{(Y)(s_r)}{1 + (k_d)(\theta_m)}, \tag{3}$$

where θ_m = sludge age, days.

Similarly, Eq. (1) also can be rearranged to show the effect of the food-to-microorganism ratio

(F:M):

$$P_x = (Y)(s_r) - \frac{(k_d)(s_r)}{(C_2)(F/M)}, \tag{4}$$

where C_2 = coefficient to match units of s_r and "F" in F:M; if s_r is BOD_5 removed, then C_2 is BOD_5 removal efficiency (about 0.9) and F:M = food-to-microorganism ratio.

The production of biological solids, P_x, decreases when θ_m increases and F:M decreases. Since biosolids handling is expensive, a lot of money have to be spent for this purpose. To overcome this problem, high values of θ_m or low values of F:M can be used to reduce the costs. Nevertheless, there are few ways to offset the cost factor that include increasing the volume of aeration tank, increasing the oxygen requirements for the aerobic biological system, etc.

2.4.2.1.2. *Effect of nitrification*

Nitrification is the biological oxidation of ammonia with oxygen into nitrite followed by the oxidation of these nitrites into nitrates. If we compare the nitrification process with other processes that are designed for carbonaceous oxidation only (BOD_5 and COD), stable nitrification processes occur at long sludge ages (θ_m) and low food-to-microorganism ratios (F:M). Furthermore, nitrification processes are always preceded by other processes that remove much of the BOD_5 and SS during the treatment process. As a result, less WAS is produced in nitrification mode compared to conventional activated sludge processes.

2.4.2.1.3. *Effect of feed composition*

The type of wastewater that is fed to the activated sludge process cause a major influence on the gross yield (Y) and decay (k_d) coefficients. Most of industrial wastes contain large amounts of soluble BOD_5 but small amounts of suspended or colloidal solids. These wastes normally have lower Y coefficients than that are obtained with domestic primary effluent.

2.4.2.1.4. *Effect of DO concentration*

Different kinds of DO levels have been maintained in an activated sludge processes. Solids production will increase at low DO concentrations such as 0.5 mg/L even when other factors are constant in conventional activated sludge systems. However, the use of pure oxygen instead of air will reduce biosolids production. This is because high DO levels accomplished through the use of pure oxygen.

2.4.2.1.5. *Effect of temperature*

The biological activity is affected by the coefficients Y (gross yield) and k_d (decay). They vary due to the temperature of the wastewater.

2.4.2.1.6. *Effect of feed pattern*

There are various feed patterns for the activated sludge process including step feeding, conventional plug-flow, contact stabilization, and complete-mix. However, for design purposes, feed pattern should be ignored when estimating solids production.

2.4.2.2. Concentration of WAS

Basically, the volume of biosolids produced by the process is directly proportional to the dry weight and inversely proportional to the thickness or solids concentration in the waste sludge stream. It was found that the values for WAS concentration can vary. In practice, the concentration of SS is ranging from 1,000 to 30,000 mg/L or 0.1–3%. The method of biosolids wasting is an important variable that can affect WAS concentration. Figure 1 illustrates a number of different methods of biosolids

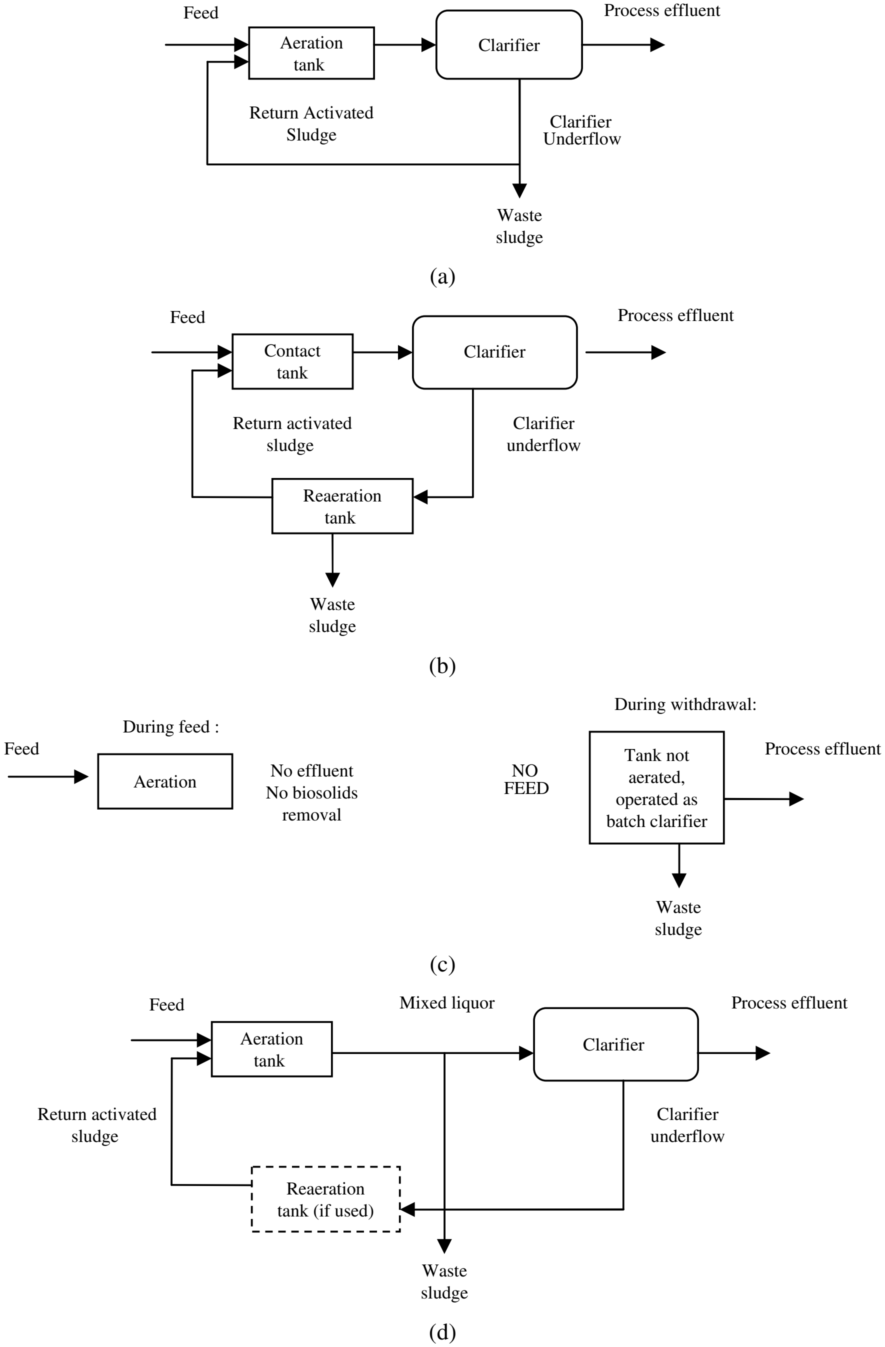


Figure 1. Biosolids Wasting Methods[5]: (a) Wasting from Clarifier Underflow, (b) Wasting from Reaeration Tank, (c) Wasting by Batch Settling, and (d) Wasting from Mixed Liquor.

wasting. From Fig. 1, it can be seen that sludge solids may be wasted from the clarifier underflow. It has been argued that wasting solids from the mixed liquor should improve the control of the process.[4] The concentration of waste sludge removed from the activated sludge process is same as the mixed liquor SS. The percentage removed is about 0.1–0.4%. This low concentration may cause a large volume of mixed liquor must be removed to obtain a given wastage on a dry weight basis. The most common arrangement for activated sludge process involves biosolids wasting from the clarifier underflow. This is because the concentration of sludge in this method is higher than in the mixed liquor.

2.4.2.3. Estimating WAS concentration

The settleability of the sludge and the solids loading rate to the sedimentation tank are the two primary factors that affect the concentration of WAS. These two factors have been considered in detail in the development of solids flux procedures for predicting the clarifier underflow concentration of activated sludge.

There are several factors that influence sludge settleability and the clarifier biosolids loading rate. These are discussed in the following sections.

2.4.2.3.1. *Biological characteristics of the sludge*

Maintenance of a particular mean sludge and F:M ratio may control the biological characteristics of the sludge. The presence of filamentous organisms in high concentrations can occur in activated sludge. The production of concentrated clarifier underflow will increase if the filamentous organisms are reduced through sludge age or F:M control.

2.4.2.3.2. *Temperature*

Basically, temperature changes is directly proportional to a clarifier underflow biosolids concentration (C_u) and inversely proportional to water density. Besides that, temperature can influence the settling properties of the biosolids.

2.4.2.3.3. *Solids flux*

The solids flux is the solids load from the mixed liquor divided by the clarifier area. Higher rates of solids flux require that clarifiers be operated at lower solids concentration.

2.4.2.3.4. *Limits of sludge collection equipment*

Some of the available biosolids collectors and pumps are not capable of smooth and reliable operation when C_u greater than 5,000 mg/L.

2.4.2.3.5. *Heavy SS in the sludge*

Higher C_u values usually obtained if raw wastewater is fed to the activated sludge process. The addition of chemicals into the wastewater for phosphorus and SS removal may similarly affect C_u. The additional solids from the removal process will also increase the solids load to the clarifiers.

2.4.2.4. Other properties of activated sludge

Table 4 summarizes the characteristics of an activated sludge. From Table 4, it is found that activated sludge contains higher amounts of nitrogen, phosphorus, and protein. However, the amount of grease, fats, cellulose, and specific gravity

Table 4. Activated Sludge Characteristics.[5]

Characteristics	Range of Values	Typical Value	Comments
pH	6.5–8	—	Can be less in high purity oxygen systems or if anaerobic decomposition begin
		5.5	Baltimore, Maryland
Heating value, Btu/lb (kJ/kg)	—	6,540 (15,200)	Increases with percentage volatile content
Specific gravity of individual solid particles	—	1.08	
Bulk specific gravity	—	$1 + 7 \times 10^{-8} \times C$	C is SS concentration in mg/L
Color	—	Brown	Some grayish sludge has been noted; activated sludge becomes black upon anaerobic decomposition
COD/VSS ratio	—	2.17	
C/N ratio	—	12.9	Baltimore, Maryland
		6.6	Jasper, Indiana
		14.6	Richmond, Indiana
		5.7	Southwest Plant, Chicago, Illinois
		3.5	Milwaukee, Wisconsin (heat-dried)
Organic carbon (% by weight of dry solids)	17–41	—	Zurich, Switzerland
	23–44	—	Four plants
Nitrogen (% by weight of dry solids, expressed as N)	4.7–6.7	—	Zurich, Switzerland
	—	5.6	Chicago, Illinois
	2.4–5.0	—	Four plants
	—	6.0	Milwaukee, Wisconsin

(*Continued*)

Table 4. (*Continued)*

Characteristics	Range of Values	Typical Value	Comments
Phosphorus (% by weight of dry solids, expressed as P_2O_5, divide by 2.29 to obtain phosphorus as P)	3.0–3.7	—	Zurich, Switzerland
	—	7.0	Chicago, Illinois
	2.8–11	—	Four plants
	—	4.0	Milwaukee, Wisconsin
Potassium (% by weight of dry solids, expressed as K_2O, divide by 1.20 to obtain phosphorus as K)	0.5–0.7	—	Zurich, Switzerland
	—	0.56	Chicago, Illinois
	—	0.41	Milwaukee, Wisconsin
Volatile solids (% by weight of dry solids,% ash is 100 minus% volatile)	61–75	—	Zurich, Switzerland
	—	63	
	62–75	—	
	59–70	—	Four plants
	—	76	Renton, Washington (Seattle Metro)
		88	San Ramon, California (Valley Community Services District)
	—	81	Central Plant, Sacramento County, California
Grease and fat (% by weight of dry solids)	5–12	—	Ether extract
Cellulose (% by weight of dry solids)		7	Includes lignin
Protein (% by weight of dry solids)	32–41	—	—

are lower. Furthermore, several types of microorganisms are also present in large numbers in activated sludge such as floc-forming bacteria (*Zoogloea*, *Pseudomonas*, *Arthrobacter*, and *Alcaligenes*), filamentous microorganisms (*Sphaerotilus*, *Thiothrix*, *Bacillus*, and *Beggiatoa*), and protozoa (ciliates and flagellates).

2.4.3. *Trickling Filter Biosolids*

Trickling filters have been widely used in WWTPs for many years. It is one of the oldest forms of dependable biological treatment for wastewaters. A trickling filter is an attached growth biological process that uses an inert medium such as rocks, gravel, slag, or plastic media to attract microorganisms that form a film or slime layer on the medium surface. A rotary or stationary distribution mechanism distributes wastewater from the top of the filter percolating it through the interstices of the film-covered medium. As the wastewater moves through the filter, the organic matter is adsorbed onto the slime layer and degraded by a mixed population of aerobic microorganisms attached on the media. The oxygen required for organic degradation is supplied by air circulating through the filter induced by natural draft or ventilation. As the slime layer thickens, it becomes more difficult for air to penetrate the layer and an inner anaerobic layer is probably formed. This slime layer continues to build until it eventually sloughs off, breaking off longer growth into the treated effluent as a biological biosolid that requires subsequent removal and disposal. Typically, a trickling filter is followed by a clarifier or sedimentation tank for the separation and removal of the sloughing.

Typically, the trickling filter effluent is usually fed to an activated sludge process when a clarifier is not used. Trickling filter microorganisms are biochemically similar to microorganisms that predominate in activated sludge systems. Consequently, solids production from trickling filters and from activated sludge systems is roughly similar when compared on the basis of mass of solids produced per mass of substrate removed.

The natural advantages of trickling filter have made it the main process available for the treatment of both municipal and industrial wastes for many years. Despite the well-known advantages, however, trickling filter has disadvantages. Table 5 summarizes the advantages and disadvantages of trickling filter.

Table 5. The Advantages and Disadvantages of Trickling Filter.[9]

Advantages

- Low energy requirements
- Low operation and maintenance requirements
- Can treat certain toxic wastes; for example, phenols
- Can withstand shock loading

Disadvantages

- Requires large area
- Expensive to construct
- Susceptible to ponding, odors, and filter flies
- Requires primary treatment
- Performance varies with weather

Table 6. Concentration of Trickling Filter Biosolids Withdrawn from Final Clarifiers.[5]

Type of Biosolids	Percentage of Dry Solids	Comments
Trickling filter, alone	5–10	Depends on solids residence time in trickling filter
	7	
	7	Low rate trickling
	3	High rate trickling filter
	3–4	
	4–7	
Trickling filter, combined with raw primary	3–6	

2.4.3.1. Concentration of trickling filter biosolids

Typically, loadings of trickling filter biosolids on the secondary sedimentation tank are low. The percentage of observed solids loads to activated sludge sedimentation tanks is normally around 5–10%. Trickling filter biosolids also have better thickening properties compared to activated sludge. Consequently, trickling filter biosolids can be withdrawn at a much higher concentration than WAS. Table 6 lists the concentration of trickling filter biosolids withdrawn from final clarifiers.

2.4.3.2. Properties of trickling filter biosolids

Table 7 shows the properties of trickling filter biosolids. The microbial population that lives in a trickling filter is complex. It includes many species of algae, bacteria, fungi, protozoa, worms, snails, and insects. Normally, filter flies and their larvae are often present in large numbers around trickling filters.

2.4.4. *Rotating Biological Contactor Biosolids*

A rotating biological contactor or RBC is a biological treatment process used in the treatment of wastewater following primary treatment. The basic biological process is similar to that occurring in the trickling filter where the RBC process allows the wastewater to come in contact with a biological medium in order to remove pollutants in the wastewater before the discharge of the treated wastewater to the environment. Basically, the RBC play an important role to remove biodegradable organic matter (BOD_5), TSS and convert ammonia-N and organic-N to nitrate in wastewater treatment.[10]

An RBC comprises of a series of closely spaced, circular, and plastic discs that are attached to a rotating horizontal shaft rotate slowly through the wastewater.

Table 7. Trickling Filter Biosolids Properties.[5]

Property	Value	Comments
Volatile content (% of total solids)	64–86	
Nitrogen (% of total solids)	1.5–5 2.9 2.0	Depends on length of storage of sludge in filter
Phosphorus as P_2O_5 (% of total solids)	2.8 1.2	
Fats (% of total solids)	6	Ether soluble
Grease (% of total solids)	0.03	Test slime grown in primary effluent
Specific gravity of individual solid particles	1.52 1.33	
Bulk specific gravity (wet)	1.02 1.025	
Color	Grayish brown Black	

About 40% of the disc area is submerged in the wastewater.[11] Bacterial growth on the surface of the discs leads to the formation of a biofilm (slime layer) that covers the wetted surface of the discs. As the RBC rotates, the attached biofilm that grows on the surface of the discs moves into and out of the wastewater. While the discs submerged in the wastewater, the microorganisms absorb organic material in the wastewater. Meanwhile, when the discs are rotated out of the wastewater, the microorganisms will absorb oxygen for aerobic decomposition. Excess biofilm shears off at a steady rate as the disc rotates. These solids are carried through the RBC system for subsequent removal in a clarifier. These removed solids are known as RBC biosolids. RBC biosolids is roughly similar in quantity by dry weight, nutrient content, and other characteristic to trickling filter biosolids.

2.4.5. *Coupled Attached-Suspended Growth Biosolids*

Generally, there are several installations of combination of coupled attached and suspended growth processes in the United States. The purpose of these dual processes is usually to treat the strong wastes or where nitrification is required. Basically, the attached growth reactor is a trickling filter or a rotating biological contactor. Its role is to reduce the load on the suspended growth process. Meanwhile, the suspended growth process uses an aeration tank and a final clarifier. Flow recirculation is usually practiced around the attached growth reactor. The biosolids produced from this process is similar to activated sludge, both in quantity and in characteristics.

2.4.6. *Denitrification Biosolids*

Denitrification is referred to biological process for the removal of nitrate from wastewater. During the treatment, an electron donor, i.e. carbon in primary effluent or methanol, is added to the nitrate-bearing wastewater. Denitrifying bacteria extract energy for growth from the reaction of nitrate with the electron donor. This can be illustrated in Eq. (5):

$$\begin{aligned}&\text{Nitrate} + \text{electron donor (reduced state)}\\&\quad\longrightarrow \text{nitrogen gas} + \text{oxidized electron donor} + \text{energy}.\end{aligned}\tag{5}$$

Denitrification has been extensively studied and a few denitrification processes have been built into municipal plants. Denitrifying bacteria can grow either in a suspended growth system similar to activated sludge or in an attached growth system similar to a trickling filter. Biosolids production for ordinary nitrified domestic waste is about 30 mg/L of wastewater treated.[12]

2.5. ***Other Wastewater Solids/Residuals***

Other than primary, chemical, and biological sludges that have been discussed in the previous section, there are several other residual solids that must be removed from the wastewater treatment process, which include screenings, grit, scum, septage, and filter backwash.[5] Although their quantities are significantly less than those of sludge in volume and weight, their removal and disposal are very important. There are several reasons to show that these wastewater solids must be handled and treated properly, which include: to obtain good quality of effluent to be discharged, to protect the environment, and to achieve reasonable treatment plant operations.

2.5.1. *Screenings*

Screenings can be defined as materials that can be removed from wastewater by screens or racks with openings of 0.01 in. (0.25 mm) or larger. They can be classified into two types, which include coarse screenings and fine screenings. Generally, coarse screenings consist of rags, string, lumber, rocks, tree roots, leaves, branches, diapers, and plastics. Quantities of coarse screenings vary from 4 to 40 mL/m^3 of wastewater. Fine screenings normally consist of SS that pass the 0.25 mm screen opening. Basically, the properties of screenings are solids content, volatile content, fuel value, and bulk wet weight. Table 8 shows all the properties that have been mentioned above.

Screens are usually installed at the entrance of the WWTP to protect mechanical equipment, avoid interference with plant operations, and prevent objectionable floating materials such as rags or rubber from entering settling tanks. There are two

Table 8. Properties of Screenings.[5]

Properties				
Solid Content, % Dry Solids	Volatile Content, %	Fuel Value, Btu/lb Dry Solids	Bulk Wet Weight, Lb/cu ft	Comments
20	—	5400[a]	60	Coarse screenings; fine screenings may have lower solids content
10–20	80–90	—	40–60	Common values
8–23	68–94	—	53–67	Various plants, fine screens, 0.03–0.12-in. openings
6.1	96	—	—	Thickened ground screenings from 0.75-in. racks; after grinding; screening were thickened on a static screen with 0.06-in openings
17	96	—	—	Dewatered ground screenings from 0.75-in. racks; after grinding, screenings were dewatered on a rotating drum screen with 0.03-in. opening
—	86	7820	—	Fine screenings

Notes: [a]Computed; 1 Btu/lb dry solids = 2.32 kJ/dry solids; 1 lb/cu ft = 16.03 kg/m^3; and 1-in. = 2.54 cm.

types of screens used to remove screenings, which are coarse screens and fine screens. The opening for coarse screens is larger than 0.25 in. (6 mm) while fine screens have opening ranging from 0.01 to 0.25 in. (0.25–6 mm). The installation of screens is usually to treat wastewater as it enters the treatment plant. Coarse screens play a significant role to remove large solids, rags, and debris from interfering with other plant equipment during wastewater treatment. Table 9 describes the various types of coarse screens.

Meanwhile, fine screens are typically used to remove material that may create operation and maintenance problems in downstream processes, particularly in systems that lack primary treatment. Fine screens may act like primary sedimentation tank although they cannot remove as much of the solids as do sedimentation tanks. Normally, this unit is protected by coarse screens that have been built upstream.

Table 9. Description of Coarse Screens.[13]

Screen Type	Description
Trash rack	• Designed to prevent logs, timbers, stumps, and other large debris from entering treatment processes • Opening size: 38–150 mm (1.5–6 in.)
Manually cleaned bar screen	• Designed to remove large solids, rags, and debris • Opening size: 30–50 mm (1–2 in.) • Bars set at 30–45° from vertical to facilitate cleaning • Primarily used in older or smaller treatment facilities or in bypass channels
Mechanically cleaned bar screen	• Designed to remove large solids, rags, and debris • Opening size: 6–38 mm (0.25–1.5 in.) • Bars set at 0–30° from vertical • Almost always used in new installations because of large number of advantages relative to other screens

The screens with 0.09–0.25 in. (2–6 mm) openings have potential to remove about 5–10% of SS from the wastewater. Meanwhile, about 25–35% of SS can be removed if 0.03–0.06 in (0.8–1.5 mm) screen openings are used.[14] Basically, the quantity of screenings produced depends on several factors such as screen opening size, shape of openings, type of sewer system, operating practices, and length of sewer system. Generally, the screenings produced should be handled in a proper manner. Screenings may be ground and handled with other sludges. Screenings are normally hauled to a landfill or incinerated at the incineration plant with the ash disposed in the landfill. Table 10 shows various methods that are used to handle screenings and also shows the advantages and disadvantages of each method. Some treatment plants return the screenings to the liquid stream after macerating or comminuting. This is not recommended because many of the downstream equipment such as mixers, air diffusers, and electronic probes are subject to fouling from reconstituted rags and strings.

2.5.2. *Grits*

Grit includes sand, gravel, cinder, or other heavy solid materials with higher specific gravity than the organic biodegradable solids in the wastewater. Besides that, grit also contain with eggshells, bone chips, seeds, coffee grounds, nails, bottle caps, and large organic particles such as food waste. These residuals must be removed from wastewater before entering primary sedimentation tank or other major processes. There are several reasons to show the importance of grit removal before any further

Table 10. Methods of Handling Screenings and Their Advantages and Disadvantages.[5]

Method	Advantages	Disadvantages
1. Comminuting within main wastewater stream; handle comminuted screenings with other wastewater solids, e.g. primary sludge	• Highly mechanized, low operating labor requirement. • Minimizes the number of unit operations. • Usually free of nuisance from flies and odors. • Widely used, familiar to plant operator.	• Sludge contains screenings, which may interfere with public acceptance for reuse of sludge as a soil amendment. • Sludge probably needs further maceration or screening if it is to be pumped or thickened in a disc nozzles centrifuge. • If sludge is to be digested, digesters must be cleaned more often. Plastics and synthetic fabrics do not decompose in digesters. Aggravates digester scum problems. Ground screenings tend to agglomerate in digesters. • Not appropriate if SS removal is required (fine screens). • Not appropriate for very large screenings loads, especially if high grit loads are also present (large plants, combined sewers).
2. Removal[a] from main stream, grinding, or maceration and return to main stream	Similar to Method 1, except more complex mechanically.	Similar to Method 1, except Method 2 can be designed for every large flows and screenings loads. Method 2 is more expansive than Method 1 for small screening loads.
3. Removal[a] from main stream, draining or dewatering, and landfill	• Keeps screening out of other sludges; avoids disadvantages of Methods 1 and 2. • Can be fairly well mechanized.	• Transport of screenings may be difficult. • Unless carefully designed and operated, causes fly and odor nuisances and health hazards. • Regulations for landfill disposal may strongly affect operations.
4. Removal[a] from main stream, dewatering, incineration, and landfill of ash	• Keeps screenings out of other sludges; avoids disadvantages of Methods 1 and 2. • Ash is very small in volume and easy to	• High cost if an incineration is required for screenings alone. • Unless incinerator is properly designed and operated, air pollution (odor and particulates) will be serious.

(*Continued*)

Table 10. (*Continued*)

Method	Advantages	Disadvantages
	transport and dispose of. • If incineration is used for other sludges and/or grit, then screenings can be added at modest cost. • Pathogen kill.	• Not well adapted to wide fluctuation in screenings quantities, unless screenings are only a small part of the total incinerator load.
5. AD of fine screenings alone (not mixed with other solids).	—	Digestion was tested but found to be impractical.
6. AD of screenings together with scum but separate from other sludges.	—	Tested at Malabar Plant, Sydney but found to be inoperable. Material handling was the chief difficulty.

[a] Mechanical removal is usually practiced at large plants. Manual removal is frequently used at small plants. The advantages of manual removal are simplicity and low capital cost; the disadvantages are high operating labor requirements and fly and odor problems. A common arrangement at small plants is to install a single comminutor with a manually cleaned bar rack as a standby unit.

treatment, which include[13]:

- to prevent unnecessary abrasion and wear of mechanical equipment,
- grit deposition in pipelines and channels, and
- accumulation of grit in anaerobic digesters and aeration basins.

Basically, the quantities of grit will vary greatly from one WWTP to other. The quantity of grit produced depends on several factors such as[5]:

- type of collection system,
- degree of sewer system corrosion,
- scouring velocities in the sewers,
- presence of joints and cracks in the sewer system,
- structural failure of sewers,
- quantities of industrial wastes,
- degree to which household garbage grinders are used,
- efficiency of grit removal at the treatment plant,
- amount of septage produced, and
- occurrence of construction in the service area or at the treatment.

Normally, the properties of grit includes moisture content, volatile content, specific gravity, bulk density, particle size, volatile solids, etc. Grit that is removed from

Table 11. Properties of Grit.[5,15]

Properties	Values
Moisture content (%)	13–65
Volatiles content (%)	1–56
Specific gravity	1.3–2.7 (can be as low as 1.3 when large amount of organic material is agglomerated with inert)
Volatile solids (%)	8–46
Bulk density (kg/m^3)	1600
Particle size (mm)	0.2 and larger

the treatment unit is normally inert and relatively dry. Table 11 summarizes the properties of grit, which have been mentioned above. The values recorded are based on the analyses carried out by previous researchers.

The production of grit in a large amount has a potential to interfere the treatment processes or can cause undue mechanical wear and abrasion. This situation can increase the maintenance on wastewater treatment equipment. To overcome these problems, grit should be handled in a proper manner. The first step in grit handling is the separation of the grit from the main stream of wastewater. Typically, grit removal facilities precede primary clarification and follow screening and comminution. The processes or devices used for grit removal all based on the fact that grit is heavier than the organic solids, which should be kept in suspension for treatment in following unit processes. Nowadays, many types of grit removal system are exist. These include aerated grit chambers, detritus tanks, vortex type grit removal systems, horizontal flow grit chambers etc.[13,16] When selecting a grit removal process, several factors must be considered that include:

- quantity and characteristics of grit,
- potential adverse effects on downstream processes,
- head loss requirements,
- space requirements,
- removal efficiency,
- organic content, and
- cost.

Moreover, the grit is separated from the primary sludge. Usually, this separation process occurs in cyclonic separator.[16] After the grit has been separated from water and putrescible materials, it is hauled to the landfill. Landfilling is found to be the most suitable method for the disposal of grit from small plants. However, grit removed is usually washed to remove organic material prior to its disposal. Generally, unwashed grit may consist 50% or more organic material.[15] If it is not washed and cleaned, it creates nuisance problem by causing foul odors, which attract rodents.

2.5.3. *Scum*

Scum is a minor component of wastewater solids collection and consists of the floatable materials skimmed from the surface of primary and secondary settling tanks, grit chambers, and chlorine contacts tanks.[4] Normally, scum may consists of varying quantities of skin, soap, grease, vegetable and mineral oils, animal fats, waxes, food wastes, vegetable and fruit skins, bits of wood, paper and cottons, cigarette tips, grit particles, and a variety of personal hygiene items such as adhesive bandages, plastic tampons applicators, and condoms. Scum usually has a specific gravity less than 1.0, which is around 0.95.[4,5]

Typically, scum production is influenced by few factors as listed below[5]:

- Wastewater temperature, dissolved solids, and pH;
- Design and operation of grease traps at commercial kitchens, gas stations, and industries;
- Amount and character of septage that is mixed with the wastewater;
- Habits and attitude of residential population and owner of small businesses;
- Preaeration and prechlorination during wastewater treatment;
- The efficiency of upstream processes in removing colloidal grease;
- Scum that is returned from sludge handling;
- Scum removal effectiveness;
- Tendency of sludge solids to float in sedimentation tanks due to formation of gas bubbles;
- Process unit from which scum is removed; and
- Actinomycete growths in activated sludge.

Basically, the properties of scum comprised of solids content, volatile content, fuel value, and grease content of scum. Table 12 summarizes the production and the properties of scum. The values recorded in the Table 12 are based on the data gathered from various treatment plants in the United State.

Scum should not be stored for more than a few days because the floating material such as grease will begin to decompose and as a result, it will produce foul odor. The floating grease coagulates at room temperature and this will normally clog pipes, pumps, and digesters. In order to eliminate clogging and adhesion problem, heated pipes are often installed during scum processing. Piping should be heated to a minimum of 15°C. However, higher temperatures are preferred if pipe sizes less than 100 mm are used.[5]

Table 12. Scum Production and Its Properties.[5]

Treatment Plant	Quantity (Volume), Gallons/Million Gallons of Wastewater	Quantity (Dry Weight)		Solids, %	Volatile Solids, % of Total Solids	Fuel Value, Btu/lb Dry Solids	Comments
		lb/Million Gallons of Wastewater	mg/L of Wastewater				
Dublin-San Ramon, California	250	—	—	—	—	—	From primary sedimentation, domestic waste
Lower Allen Township, near Harrisburg, Pa	14	31	4	25[a] 27[a]	46[a] 42[a]	6,900[a] 3,100[a]	From low lime primary sedimentation (pH 9.4–9.8), after heated thickener
Northwest Bergen County	25	19	2.3	9	—	—	From gravity thickener
Wichita, Kansas	9	—	—	—	—	—	From primary sedimentation. Grease is 30% of skimmings after decanting
	0.7	—	—	—	—	—	Grease balls from preaeration tanks
Minneapolis-St Paul, Minnesota	—	—	—	—	98	13,000	From primary sedimentation
East Bay, Oakland, California	19	82	9.8	54	96	—	Average, July 1969–June 1970[b]
	—	110	13	64	99	—	Maximum month
	—	60	7.2	43	81	—	Maximum month
	29	—	—	51	—	14,000	1965–1966 data
West Point, Seattle, Washington	50	24	2.9	6	—	—	As pumped from primary sedimentation tanks
	8	19	2.3	30	—	—	As above, after decanting, 6.4% grease
	130	—	—	—	—	—	From sedimentation tanks under poor condition[c]

(*Continued*)

Table 12. (*Continued*)

Treatment Plant	Quantity (Volume), Gallons/Million Gallons of Wastewater	Quantity (Dry Weight) lb/Million Gallons of Wastewater	Quantity (Dry Weight) mg/L of Wastewater	Solids, %	Volatile Solids, % of Total Solids	Fuel Value, Btu/lb Dry Solids	Comments
San Mateo, California	—	95	11	—	—	—	—
Salisbury, Maryland	200	—	—	—	—	—	From primary clarifiers. Heavy grease load from industry
Three New York City plants	0.3–5	1.2–15	0.1–2	40–52	—	—	From primary clarifiers; about 80% of solids are grease
Jamaica, New York City	3	10	1.2	48	—	—	From secondary clarifiers (no primary)
County Sanitation Districts, Los Angeles County, CA	—	87	10	—	—	—	Primary sedimentation
Albany, Georgia	3,000	140	17	0.57	—	—	Primary sedimentation. Heavy industrial load

[a] Two samples. About 50% of nonvolatile solids was calcium carbonate.

[b] 91% of total solids were oil and grease. Scum from primary sedimentation, measured after decanting in a heated unit.

Sludge was tending to float in the sedimentation tanks. Amount shown is estimate of pumpage. Skimming system was unable to keep up with scum production under these poor conditions.

1 gallon/million gallons = $m^3/L \times 10^6\ m^3$, 1 lb/million gallons = $0.12\ kg/L \times 10^3\ m^3$, and 1 Btu/lb dry solids = 2.32 kJ/kg.

The removal of scum is an important function of primary treatment. The presence of these materials can increase the organic load and it also might cause various operational problems to downstream treatment processes.[17] The collection and removal of scum is required for all settling tanks. The removal of scum from the tank will take place immediately ahead of the outlet weirs. The equipment used may be automatic or manually operated. There are several methods that can be used to remove scum, which include: (1) manually up an inclined apron, (2) with a horizontal, slotted pipe that can be rotated by a lever or a screw, (3) by a transverse rotating helical wiper, (4) with chain and flight collectors, and (5) with cum rakes where bridge-type sedimentation tank equipment is used.[15] The collected scum is usually discharged to the digesters and disposed of with the sludge produced at the plant. However, many plants have already used separate system for scum disposal nowadays. There are several methods used in scum disposal including digestion, landfilling, and incineration. Table 13 summarizes the method of handling scum and their advantages and disadvantages of various approaches to scum disposal.

Table 13. Methods of Handling Scum and their Advantages and Disadvantages of Various Approaches to Scum Disposal.[5]

Method	Advantages	Disadvantages
1. Mix with other sludges, digest aerobically	• Partial decomposition occurs, so some of the scum does not require further handling. • Avoid complexity of separate handling. • Widely used.	• May cause the formation of grease balls, which must be manually removed and disposed of, and which may increase odors. • May cause petroleum contamination of sludge, which will interfere with reuse. • Degrades appearance of sludge if to be reused. • May cause scum buildup due to return of scum containing liquors from sludge handling to influent wastewater.
2. Mix with other sludges, digest anaerobically	Similar to Method 1.	• If digester is not strongly mixed, greatly increases cleaning requirement. Digester cleaning is expansive and odorous; also material still requires disposal.

(*Continued*)

Table 13. (*Continued*)

Method	Advantages	Disadvantages
		• Even if digester is well mixed, a scum blanket will form to some extent; therefore, digester must be physically larger. • Degrades appearance of sludge if to be reused; may cause petroleum contamination. • May cause scum buildup. • Requires good decanting to avoid pumping excess water to the digester.
3. Landfill separately	Low capital cost.	• May have very high operating cost. • Possible odors during storage. • Requires good decanting to minimize volume and fluidity of scum.
4. Burn in open lagoon	Very low cost.	Severe air pollution.
5. Incinerate in separate "Watergrate" furnace	Very small amount of ash in slurry.	• High capital cost, especially for small plant. • Despite low emissions, may not be acceptable to air pollution regulators. • Problems with feed systems.
6. Incinerate in separate single purpose multiple-hearth furnace	Very small amount of ash.	• High capital cost. • High maintenance cost. • Despite low emissions, may not be acceptable to air pollution regulators. • Requires good decanting
7. Incinerate in multiple-hearth furnace with other wastewater solids	• Low incremental cost. • Fuel value of scum can be used to offset fuel requirement of other solids.	• Requires good decanting • Requires very careful feed to the furnace; otherwise causes high maintenance and severe smoke problems. These problems can be avoided.
8. Incinerate in fluidized bed furnace with other wastewater solids	Similar to Method 7.	• Unless well decanted, can tax furnace capacity. • If scum is mixed with sludge before injection into furnace, unstable operation is likely.

(*Continued*)

Table 13. (*Continued*)

Method	Advantages	Disadvantages
9. Reuse for cattle food	• Provide reuse, not disposal. • Low capital cost.	• Toxic organic materials (e.g. DDT) tend to concentrate in grease. • Treatment for reuse must begin within a few days; otherwise grease begin to decompose. • Requires good decanting because of long distance transportation. • Subject to interference from actinomycete growths in activated sludge, which increases the amount of solids that are not grease but are in the scum.
10. Reuse for low-grade soap manufacture	Same as Method 9.	• Similar to Method 8, but less serious. • Caustic soda could be added at the treatment plant, preventing decomposition and probably making the material more usable to grease reclaimers, but raising operating costs.
11. Return to influent wastewater	• Almost zero direct cost. • Highly suitable for scum from chlorine contact tanks, secondary clarifiers, etc. when scum is removed from primary sedimentation tanks.	• Slight increase in hydraulic load on the treatment plant. • Inapplicable to the main source of scum (primary sedimentation tanks if used, secondary clarifiers if primary tanks are not used).

2.5.4. *Septage*

Septage can be defined as a liquid and solid material pumped from a septic tank, cesspool, or other treatment facility after it has accumulated over a period.[18] Generally, septage consists of all the household wastes that are disposed of through a home's plumbing system that neither drain out into the soil nor are converted to gases by the special bacteria that are in the tank. Normally, household wastes derive from the toilet, bath or shower, sink, garbage disposal, dishwasher, and washing machine. Septage may also contain the pumpings from the septic tanks of schools, motels, restaurants, etc.[5]

Septage is generally split into three parts in a septic tank, which are:

(i) *Scum* — which floats at the top and is generally where the bacteria live that treat the waste.
(ii) *Effluent* — which is the semi-treated liquid that comprises the majority of the material in the septic tank.
(iii) *Sludge* — the solids that settle at the bottom of the tank.

A septic tank will usually retain 60–70% of the solids, oil, and grease that enter it. Meanwhile, 20–50% of the total septic tank volume represents the sludge that settles at the bottom of the tank. Basically, the handling and disposal of septage are based on the characteristics and volume of septic waste. The characteristics of septage consists of total solids (TS), total volatile solids (TVS), total suspended solids (TSS), volatile suspended solids (VSS), biochemical oxygen demand (BOD), chemical oxygen demand (COD), total Kjeldahl nitrogen (TKN), ammonia nitrogen (NH_3), phosphorus, alkalinity, grease, pH, total coliform, and fecal coliform. Septage characteristics for conventional parameters, metal, and organic content are presented in Table 14.

There are a few factors that affect the physical characteristics of septage including[18]:

- climate,
- user habits,
- septic tank size, design, and pumping frequency,
- water supply characteristics and piping material,
- the presence of conservation fixtures and garbage disposals, and
- the use of household chemicals and water softener.

Septage requires special handling, treatment, and disposal. This is because septage has an offensive odor and appearance besides containing significant levels of grease, grit, hair, and debris. These residuals also have a tendency to foam upon agitation and a resistance to settling and dewatering. Besides that, septage also has a potential as a host for many disease-causing organisms such as viruses, bacteria, and parasites. The major reason for providing adequate treatment and disposal systems is to protect public health and also the environment.

Basically, the basic methods for treating and disposing of septage are by land application, treatment at WWTPs, and treatment at independent septage treatment plant.[19] Table 15 summarizes the methods used for septage treatment and management, various options for each methods and their advantages and disadvantages. Basically, there are few factors that influence the process of selection, which include land availability and site conditions, buffer zone requirements, hauling distance, fuel costs, labor costs, costs of disposal, and other legal and regulatory requirements.

Table 14. Septage Characteristics for Conventional Parameters, Metal and Organic Content.[18,19]

	Concentration (mg/L)		
Parameter	Average	Minimum	Maximum
Conventional parameters			
Total solids	34,106	1,132	130,475
Total volatile solids	23,100	353	71,402
Total SS	12,862	310	93,378
VSS	9,027	95	51,500
BOD	6,480	440	78,600
COD	31,900	1,500	703,000
TKN	588	66	1,060
Ammonia nitrogen	97	3	116
Total phosphorus	210	20	760
Alkalinity	970	522	4,190
Grease	5,600	208	23,368
pH	—	1.5	12.6
Total coliform		10^7/100 mL	10^9/100 mL
Fecal coliform		10^6/100 mL	10^8/100 mL
Metals			
Iron	39.3	0.2	2,740
Zinc	9.97	< 0.001	444
Manganese	6.09	0.55	17.1
Barium	5.76	0.002	202
Copper	4.84	0.01	261
Lead	1.21	< 0.025	118
Nickel	0.526	0.01	37
Chromium (total)	0.49	0.01	34
Cyanide	0.469	0.001	1.53
Cobalt	0.406	< 0.003	3.45
Arsenic	0.141	0	3.5
Silver	0.099	< 0.003	5
Cadmium	0.097	0.005	8.1
Tin	0.076	< 0.015	1
Mercury	0.005	0.0001	0.742
Organics			
Methyl alcohol	15.8	1	396
Isopropyl alcohol	14.1	1	391
Acetone	10.6	0	210
Methyl ethyl ketone	3.65	1	240
Toluene	0.17	0.005	1.95
Methylene chloride	0.101	0.005	2.2
Ethylbenzene	0.067	0.005	1.7
Benzene	0.062	0.005	3.1
Xylene	0.051	0.005	0.72

Table 15. Septage Treatment and Disposal Methods, their Advantages, and Disadvantages.[19]

Method and Option of Treatment	Description	Advantages	Disadvantages
Treatment at independent septage treatment plant *Options*: • Stabilization lagoon • Chlorine oxidation • Aerobic digestion • Anaerobic digestion • Biological and chemical treatment • Conditioning and stabilization • Composting	A facility is constructed solely for the treatment of septage. Treatment generates residuals that must be disposed of.	Provides regional solution to septage management.	• High capital and operation and maintenance costs. • Requires high skill levels for operation.
Treatment at wastewater treatment plants *Options*: • Addition to upstream sewer manhole • Addition to plant headworks • Addition to sludge handling process • Addition to both liquid stream and sludge handling processes	Septage is added to the plant headworks, upstream manhole or sludge handling process for co-treatment with sewage or sludge. Septage volumes that can be accommodated depend on plant capacity and types of unit processes employed.	• Most plants are capable of handling some septage. • Centralizes waste treatment operations.	• Potential for plant upset if septage addition not properly controlled. • Increased residuals handling and disposal requirements.
Land application *Options*: • Surface application • Subsurface incorporation • Burial	Septage is applied to sites infrequently visited by the public. Stabilization to reduce odors, pathogens, and vector attraction may be encouraged or required by the state. Land application may be by hauler truck or other vehicle to apply septage to the land surface or by specialized equipment to inject septage beneath the soil surface.	• Simple and economical. • Recycle organic material and nutrients to the land. • Low energy use.	• Need for holding facility during periods of frozen or saturated soil. • Need for relatively large and remote land area.

2.5.5. *Filter Backwash*

Filters are normally referred to single-medium filters (sand filters), dual and mixed media filters, and microstrainers.[5] Single-medium filters only have one type of medium. In these filters, usually sand or anthracite coal are used for filtration. Meanwhile, dual-medium filters have two types of mediums that are crushed anthracite combined with sand. Mixed media filters are also known as multimedia filters, which consist of three types of media that include crushed anthracite, sand, and garnet.[20] The use of filters can produce very high-quality effluent, which is free from pathogens, taste, and odor without the addition of any chemical aids. Filters become clogged with floc after some time in use and they are then backwashed to remove the floc. This backwash water is discharged into settling tanks, so that the floc can settle out.

Meanwhile, microstrainers are self-contained and self-standing units that are normally installed in a concrete tank. The drum operates submerged in the flowing wastewater to approximately 2/3 of its depth. These units comprised of a rotating cylindrical drum that is made of stainless steel and supported on a steel framework. The walls are made up of a series of fine mesh windows. The wastewater influent enters the drum at one end and filters outwards through the mesh. The solid particles are retained inside the drum, as the drum rotates. Then, the solids are washed off by a jet of water. The backwash water is collected in a hopper and recycled back to the treatment plant.

Generally, in the treatment plant, wastewater is filtered to remove SS that are floating on the surface. Solids from wastewater are normally accumulated in the filters during the removal process. Then, they are removed from the filters by backwashing. During backwashing, the volume of water used is great, which is about several percentage of total wastewater flow. However, the quantity of SS in backwash is normally around 0.03–0.15%. The dry weight load is usually small compared to those from primary, chemical, and biological biosolids. Normally, backwash is returned to the influent wastewaters and their SS are removed in further wastewater processes such as primary sedimentation and activated sludge.

3. Biosolids Treatment and Processing

Most biosolids undergo additional treatment at the treatment plant before they are used or disposed of to meet regulatory requirements that protect public health and the environment, facilitate handling, and reduce costs. Only biosolids that meet certain regulatory requirements for pathogens, vector attraction reduction and metal content, can be land applied or used as compost. Even those biosolids that are disposed of rather than land applied must meet regulatory requirements. In addition, with regard to handling and cost, the water content of biosolids can affect many aspects of biosolids management, such as transportation and the size of treatment and use or disposal operations. Some biosolid treatment processes reduce the volume or mass of the biosolids (such as biosolids digestion processes). Meanwhile, other biosolids treatment and processing can increase biosolids mass (for example, lime addition to control pathogens.)

There are four major biosolids treatments and processings that occur at the WWTPs, which are discussed in this chapter; they are thickening, stabilization, conditioning, and dewatering.[1]

3.1. *Thickening*

Thickening is the process where the biosolids are condensed to produce a concentrated solids product and relatively solid-free supernatant.[21] It is commonly used as the first step in a biosolid treatment system. This process plays a significant role in order to reduce the volume of residuals, improves operation, and reduces costs for subsequent storage, processing, transfer, end use or disposal. For example, thickening from 1% to 2% solid concentration reduces the volume of biosolids by 50%, whereas increasing the solid content from 1% to 5% reduces the volume up to 80%.[1]

Generally, the processes that are commonly used to thicken biosolids are gravity thickening, floatation thickening, centrifugal thickening, gravity belt thickening, and rotary drum thickening. In addition, thickening offers the advantages such as biosolids blending, biosolids flow equalization, biosolids storage, grit removal, gas stripping, and clarification.[5]

3.1.1. *Gravity Thickening*

3.1.1.1. General description

Gravity thickening is the most common process in use at WWTPs for the concentration of biosolids before any further treatment such as digestion and/or dewatering.[1,24] The process is simple and is the least expensive of the available thickening processes. Thickeners can contribute to the upgrading of biosolids handling facilities such as increasing the hydraulic capacity of overloaded digesters or subsequent biosolids handling units. Apart from that, it can improve primary clarifier performance by providing continuous withdrawal of biosolids thereby ensuring maximum removal of solids.

Generally, gravity thickening uses the natural tendency of higher-density solids to settle out of liquid to concentrate the solids. Gravity thickeners consist of a circular tank that usually has a conical bottom. The tank is fitted with collectors or scrapers at the bottom. At first, the solids are fed into the tank through a center well, which releases the solids at a low velocity near the surface of the tank. The solids will settle

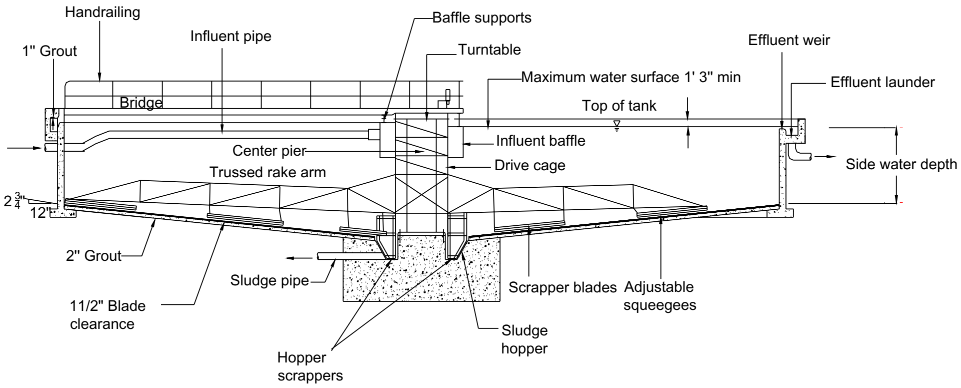
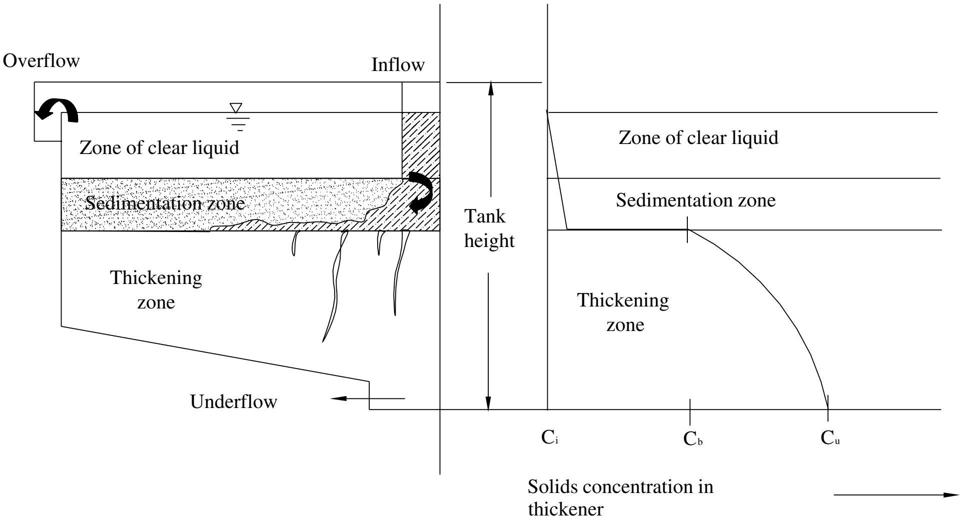


Figure 2. Schematic Diagram of Gravity Thickener.[21]

to the bottom of the tank by gravity. At the same time, the scrapers play their role to move the settled and thickened solids to a discharge pipe at the bottom of the tank. The function of a V-notch weir, which is located at the top of the tank, is to allow the supernatant to return to a clarifier.[21] Figure 2 illustrates a typical circular gravity thickener used in the gravity thickening process. In addition, many systems also use a skimmer to collect and remove any floating materials and grease that have accumulated at the surface of the tank.

Figure 3 depicts a typical solid concentration profile for wastewater biosolids in a gravity thickener. The biosolids that are discharged into the thickener disperse in the sedimentation zone. Due to a gravitational force, some parts of biosolids flow as density current to the bottom of the sedimentation zone. The solid phase creates flocs that settle on top of the thickening zone. Generally, the thickened biosolids produced in the thickening zone are transferred to downstream processing operators such as to the digesters or dewatering equipment.[4] Meanwhile, the clarified supernatant (from zone of clear liquid) is returned to the WWTP or to the primary clarifier for reprocessing.[1]

Basically, the concentration and thickening of biosolids that occur in the gravity thickener are accomplished through three different settling processes. They are gravity settling, hindered settling, and compaction settling. Gravity settling normally occurs when solid particles move downward due to their weight and attraction to the gravitational force. Settlement continues as solids begin to concentrate near the bottom of the tank. However, at this stage, the settlement rate decreases as the solids concentrations increase. This condition is known as "hindered settling." This phenomenon is influenced by the particle size distribution, density, concentration, agglomeration, as well as the hydraulic conditions in the settling zone. Furthermore, compaction settling occurs when bottom solids are further concentrated due to the pressure of solids from the upper part.[21]



C_i — inflow solids concentration
C_b — lowest concentration at which flocculant suspension is in the form of porous medium
C_u — underflow concentration from gravity thickener

Figure 3. Schematic Diagram of Typical Concentration Profile of Wastewater Biosolids in a Gravity Thickener.[3]

3.1.1.2. Advantages and disadvantages

Gravity thickeners became the most commonly used biosolids concentrating device. Their use is being challenged by other thickening processes. The advantages and disadvantages of gravity thickener are listed in Table 16.

3.1.2. *Centrifugal Thickening*

3.1.2.1. General description

The centrifugation process has been successfully used for many years in industry for separating liquids of different density, thickening slurries, and removing solids. Only now, they have been installed for regular use in WWTPs although the potential value of centrifuges for wastewater treatment has been recognized for quite some time. There are several factors that affect the increased use of centrifuges in wastewater treatment field. These include[24]:

- The improvements in centrifuge design;
- The availability of reliable performance data; and
- The advantages of centrifuges in certain instances compared to other biosolids processing facilities.

Table 16. Advantages and Disadvantages of Gravity Thickeners.[21]

Advantages

- Simple to operate and maintain.
- It has lower operating costs compared to other thickening methods such as DAF, gravity belt, or centrifuge thickening.
- Provides greatest biosolids storage capabilities.
- Requires the least operational skills.

In addition, the facilities that land apply liquid biosolids can benefit from thickening in several ways, as follows:

- Truck traffic at the plant and the farm site can be reduced.
- Trucking costs can be reduced.
- Existing storage facilities can hold more days of biosolids production.
- Smaller storage facilities can be used.
- Less time will be required to transfer solids to the applicator vehicle and to incorporate or surface apply the thickened solids.

Disadvantages

- Contributes to the production of odors.
- Grease may build up in the lines and cause a blockage.
- Septic conditions will generate sulfur-based odors.
- Requires largest area for gravity thickener equipment compared to DAF, gravity belt, or centrifuge thickener.
- Solids concentrations in the thickened solids are usually lower than for a DAF, gravity belt or centrifuge thickener.

In the late 1930s, the concept of using centrifuges for thickening municipal wastewater biosolids was first introduced in the United States. Centrifuges are used both to thicken and to dewater biosolids. Centrifugal thickening is a high-speed process that uses the force from fast rotation of a cylindrical bowl to separate wastewater solids from liquid.[22,23] During this process, the biosolids (cake) and liquid (centrate) are separated into two distinct layers. This separation occurs because of the differences in density between the solids and liquid. These products are then separately discharged from the thickener. Their application in thickening is limited, normally to WAS.[4]

At present, there are three types of centrifugal thickeners available commercially, which include solid bowl decanter, imperforate basket, and disc nozzle centrifuge. Solid bowl decanter and imperforate basket are differentiated by the method of sludge feed, the magnitude of applied centrifugal force, the method of solids and liquid discharge, cost, and performance.[23] Meanwhile, disc nozzle centrifuge has been used for thickening WAS, but does not produce a dewatered material. Table 17 lists some of the advantages and disadvantages of these three types of centrifugal thickeners.

Table 17. Advantages and Disadvantages of Centrifugal Thickeners.[22]

Advantages

- Can treat biosolids to varying degrees of thickness.
- Offer lower overall operation and maintenance costs and can outperform conventional belt filter presses.
- Require a small amount of floor space relative to their capacity.
- Require minimal operator attention when operations are stable.
- Operators have low exposure to pathogens, aerosols, hydrogen sulfide, or other odors.
- Easy to clean.
- Can handle higher than design loadings and the percentage solids recovery can usually be maintained with the addition of a higher polymer dosage.
- Major maintenance items can be easily removed and replaced.

Disadvantages

- Have high power consumption and are fairly noisy.
- Experience operating the equipment is required to optimize performance.
- Performance is difficult to monitor because the operator's view of centrate and feed is obstructed.
- Special structural considerations must be taken into account. As with any piece of high-speed rotary equipment, the base must be stationary and level due to dynamic loading.
- Spare parts are expensive and internal parts are subject to abrasive wear.
- Start-up and shut-down may take an hour to gradually bring the centrifuge up to speed and slow it down for clean out prior to shut down.

3.1.2.2. Solid bowl centrifuge

The solid bowl centrifuge also called decanter or scroll centrifuge is a continuously operating unit. In the mid 1930s, the first solid bowl decanter centrifuges to operate on wastewater biosolids was installed in the United States.[5] It consists of a rotating horizontal cylindrical bowl containing a screw type conveyor or scroll. The wastewater solids enter the cylindrical bowl through a feed pipe mounted at one end of the centrifuge. Centrifugal forces concentrates the biosolids against the bowl wall and the rotating conveyor transports the dewatered biosolids along the bowl wall to the conical section where it is discharged continuously. Meanwhile, at the other end of the centrifuge, the centrate is discharged continuously and recycled back to the WWTP.[1] Basically, there are three critical system design variables should be considered for the solid bowl decanter centrifuge, which are[23]:

i. bowl speed,
ii. pool volume, and
iii. conveyor speed.

Figure 4 illustrates the schematic diagram of a solid bowl decanter centrifuge. Table 18 lists the current advantages and disadvantages of solid bowl decanter

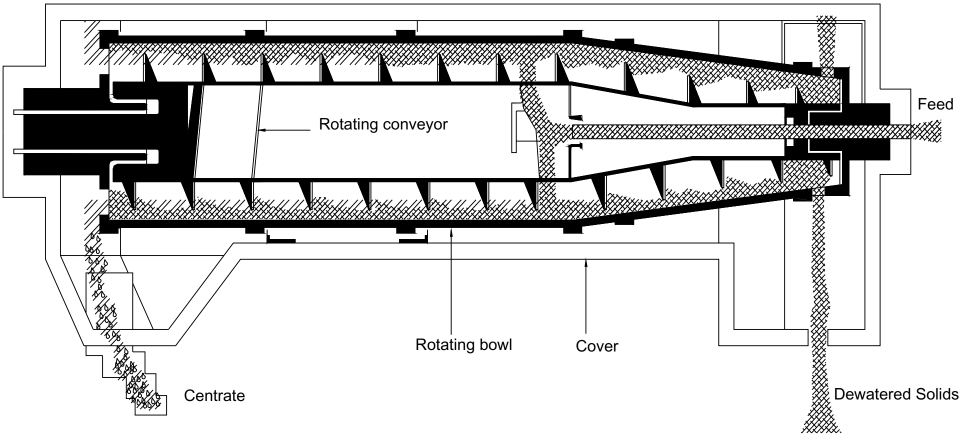


Figure 4. Solid Bowl Centrifuge.[5]

Table 18. Advantages and Disadvantages of Solid Bowl Centrifuge.[23]

Advantages
• Clean appearance, little to no odor problems, and fast start-up and shut-down capabilities.
• Yields high throughput in a small area.
• Does not require continuous operator attention.
• Easy to install and requires a relatively small area.
• Can operate with a highly variable feed solids concentration on many biosolids types.
• Can be operated either for thickening or dewatering.
• High rates of feed per unit, thus reducing the number of units required.
• Use of low polymer dosages when compared to other type of thickeners.
• Can handle higher than design loadings with increased polymer dosage, although cake solids content may be reduced.
• Quiet.
• Has low capital cost for installation.
• Has ability to constantly achieve 4–6% solids in the thickened biosolids.
Disadvantages
• Scroll wear can be a high maintenance item. In reducing wear, hardsurfacing, and abrasion protection materials are extremely important.
• Prescreening or a grinder in the feed stream is recommended.
• Requires skilled maintenance personnel in large plants where scroll maintenance is performed.
• Vibration must be accounted in designing electronic controls and structural components.
• A condition such as poor centrate quality can be easily overlooked since the process is fully contained.
• Requires extensive pretesting to select correct machine settings before placement in normal service.

Table 19. Common Design Shortcomings of Solid Bowl Decanter Centrifuge.[23]

Shortcomings	Resultant Problems	Solution
Improper materials used for scroll tips	Excessive wear	Replace with harder, more abrasion-resistant tips
Inability to remove bowl assembly during maintenance	Bowl is bulky and heavy. It cannot be removed without using lifting equipment	Install overhead crane
Rigid piping used to connect feed pipe to centrifuge	Cracked or leaking pipes or pipe connections	Replace with flexible connections
Grit present in biosolids	Excessive centrifuge wear	Install a degritting system on the biosolids or on the wastewater before biosolids removal
Electronic controls, structural components, and fasteners not designed for vibration	Electrical connections become loose, structural components and fasteners fail	Isolate sensitive electronic controls from vibration, redesign and construct structural components and fasteners to resist vibration
Electrical control panels located in same room with centrifuges, conveyor belts, etc.	Corrosive atmosphere deteriorates controls	Redesign and relocate controls in separate room away from corrosive atmosphere

centrifuges in WAS thickening. Meanwhile, the common design shortcomings associated with solid bowl are presented in Table 19.

3.1.2.3. Imperforate basket centrifuge

The imperforate basket centrifuge is a semi-continuous feeding and solid discharging unit that rotates about a vertical axis.[23] At the early stage of the centrifugation process, biosolids is fed into the bottom of the basket through the feed pipe. The biosolids form a cake on the bowl walls as the centrifuge rotates. The centrate is displaced over a baffle or weir at the top of the thickener. Biosolids feed is either continued for a preset time or until the SS in the centrate reach a preset concentration. The basket speed begins to decelerate when the sludge solids have filled the basket and the sludge feeding is stopped. At this stage, a special skimmer nozzle plays a role to skim the soft sludge on the inner circumference of the sludge mass. Typically, these skimmings are recycled back to the digesters. After skimming operation, the centrifuge decelerates further and a plowing knife cuts the sludge away from the walls. The sludge cake then drops and is discharged from the bottom of the basket. After plowing stops, the centrifuge starts to accelerate and feed sludge is again introduced. Basically, there are a few critical system design variables that should be

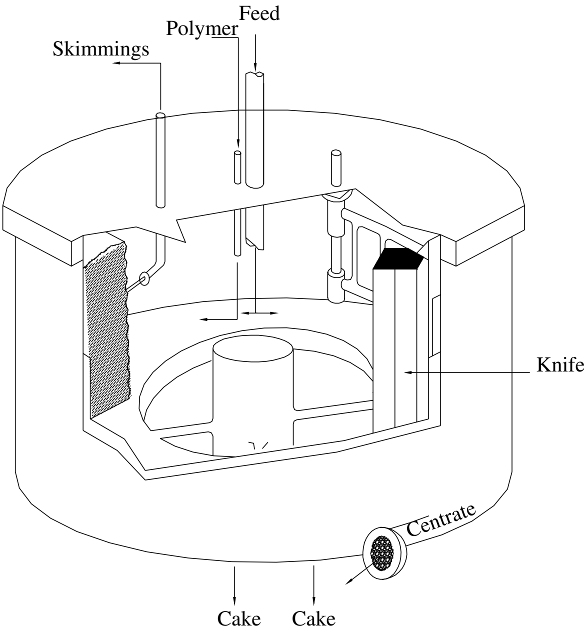


Figure 5. Imperforate Basket Centrifuge.[5]

considered for the imperforate basket centrifuge, which include:

- bowl speed,
- cycle feed time,
- skimmer nozzle rate, and
- skimmer nozzle dwell time.

Figure 5 is a schematic diagram of basket centrifuge where it illustrates the location of sludge feed inlet, polymer feed, centrate piping, and location of cake discharge.

The basket centrifuge is most commonly used for thickening WAS. Advantages and disadvantages of an imperforate basket centrifuge are presented in Table 20. Meanwhile, common design shortcomings experienced in basket centrifuge installations are listed in Table 21.

Typically, the performance of a basket centrifuge is measured by several parameters such as.[23]

- the cake solid content,
- solid capture,
- required polymer dosage, and
- the average feed rate or solid throughput.

Table 20. Advantages and Disadvantages of Imperforate Basket Centrifuge.[5,23]

Advantages

- Same machine can be used for both thickening and dewatering process.
- Very flexible in meeting process requirements.
- Not affected by grit.
- Requires little operator attention.
- Clean looking and has little to no odor problems as compared to gravity and DAF thickener.
- Excellent for dewatering hard to handle biosolids, although sludge cake solids are only 10–15% for digested primary plus WAS.
- Flexibility in producing different cake solids concentrations because of skimming ability.
- Has the lowest operation and maintenance requirements compared to other types of centrifuges.

Disadvantages

- Unit is not continuous feed and discharged.
- Requires special structural support, much more than a solid bowl centrifuge.
- Has a high ratio of capital cost to capacity.
- Discharge of wet sludges can occur if there is a machine malfunction or if the sludge is improperly conditioned.
- Provision should be made for noise control.
- Continuous automatic operation requires complex controls.

Table 21. Common Design Shortcomings of Basket Centrifuge Installations.[23]

Shortcomings	Resultant Problems	Solution
Rigid piping connections to centrifuge	Cracked or leaking pipes	Use flexible connectors
Inadequate structural support	Cracks in supports	Redesign and reconstruct
Inadequate solids capture due to insufficient machine capacity or no provision for polymer feed	High solids content in centrate	Add more machines or properly condition biosolids
Electrical control panels located in same room with centrifuges, conveyor belts, etc.	Corrosive atmosphere deteriorates controls	Redesign and relocate controls in separate room away from corrosive atmosphere
No provision for centrate sampling	Process control is impeded	Install sample tap in the centrate line
No flow meters on biosolids feed line	Process control is hindered	Install flow meters

Cake solid concentration must be considered as an average solid content. This is because solid content is maximum at the bowl wall and decreases toward the center of the basket.

The application of a basket centrifuge is suitable for small plants with capacities in the range of 1–2 millions gallons per day(MGD) (44–88 L/s) where thickening is required before and after stabilization or where dewatering to 10–12% solids is adequate.[5,23] The ability to be used for either thickening or dewatering is an advantage of the basket centrifuge.

3.1.2.4. Disc nozzle centrifuge

In 1937, disc nozzle centrifuge was first used in the United States.[5] Basically, the disc nozzle centrifuge consists of several conical-shaped discs stacked one upon another, enclosed within a rotor bowl that rotates about vertical axis. There are numerous cylindrical discharge nozzles that are placed around the periphery of the rotor bowl.[25] A typical disc nozzle centrifuge is depicted in Fig. 6.

During the thickening process, the sludge is fed to the bottom of the spinning rotor continuously. The sludge normally enters through the top; however, bottom feed is also possible. The centrifugal forces will move the heaviest solid particles directly toward the circumference of the rotor. Meanwhile, the lighter solid particles will flow through the spaces between the discs. Normally, the disc spacing that is around 1.3 mm acts to minimize settling distance. These lighter particles will

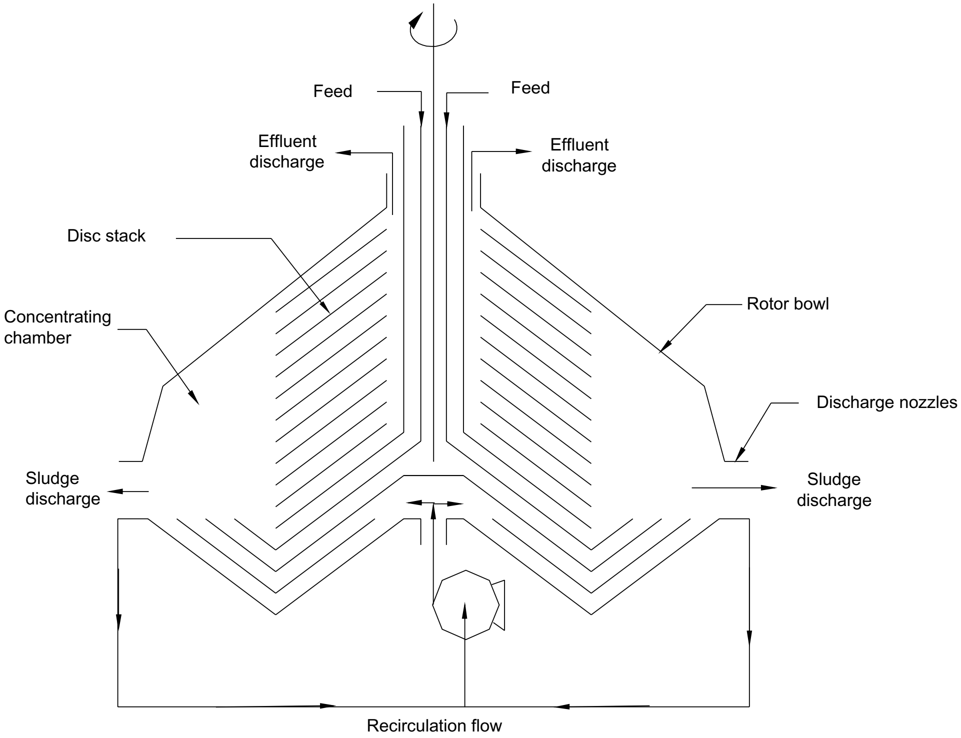


Figure 6. Disc Nozzle Centrifuge.[5]

settle on the underside of the discs and accumulate until their mass is great enough to force them to the periphery of the disc. After that, these particles will move to the edge of the rotor bowl. The centrate (clarified liquid) flows through the disc stack into the overflow chamber and is then discharged through the effluent discharge. Solid particles collected are discharged through small cylindrical nozzles. The function of these nozzles is to reduce the solids discharge rate where this will increase the retention time and agglomeration of particles in the concentrating chamber.

A portion of this concentrated sludge is discharged as the thickened product while another portion will be returned back to the base of the rotor and pumped back into the concentrating chamber. This recirculation offers a few advantages such as:

- increases the overall underflow concentration,
- minimizes particle agglomeration inside the rotor by flushing action, and
- helps to achieve a stable separation equilibrium.

Table 22 lists the advantages and disadvantages of disc nozzle centrifuge.

Generally, the application of disc nozzle centrifuges are only suitable to sludge consisting of smaller particles that are less than 400 μm and void of fibrous material. For wastewater treatment, disc nozzle centrifuge can only be equipped at the plants that have the primary treatment and have already separated the primary sludge from the waste activating sludge. Generally, only activated sludge can be thickened by using this method.

Basically, there are a few critical system design variables that should be considered for the disc nozzle centrifuge, which include[5]:

- bowl diameter,
- bowl speed,
- operation of recycle,

Table 22. Advantages and Disadvantages of Disc Nozzle Centrifuge.[5]

Advantages
• Produces highly clarified centrate without the use of chemicals.
• Has large liquid- and solid-handling capacity in a very small space.
• Produces little or no odor problem.
Disadvantages
• Can only be used on sludge with particle sizes of 400 μm or less.
• Requires extensive prescreening and grit removal.
• Requires relatively high maintenance if pretreatment system is improperly designed.
• Requires skilled maintenance personnel.

- disc spacing, and
- nozzle configuration.

However, it was found that the most important consideration for the performance of disc nozzle centrifuge is the nature of sludge. Same with other centrifuge applications, an increasing sludge volume index (SVI) influences the performance of thickeners. The concentration of the thickened sludge tends to increase with the increasing solids concentration in the inlet. Thickened sludge will be 5–10 times more concentrated than the feed. The capability to concentrate will decrease as the inlet solids become more concentrated.[5]

3.1.3. *Floatation Thickening*

Generally, floatation is known as a process used for separating solid particles from a liquid phase or water when air is introduced into the water.[1,5] The introduction of air into the water will produce fine bubbles either adhere to or are absorbed by the solids that are then lifted to the surface. Normally, the floatation process separates the particles with a higher density than the liquids. In wastewater treatment, floatation thickening is used most efficiently for waste sludge from suspended growth biological treatment processes (activated sludge process) or the suspended growth nitrification process where solids contents of 4% or higher are obtained.[4,26] In a meantime, this method also applicable for thickening other types of sludge such as primary sludge, trickling filter humus, aerobically digested sludge, and sludge containing metal salts from chemical treatment.

Typically, there are three basic variations of the floatation thickening operation, which are dissolved air floatation (DAF), dispersed air floatation, and vacuum floatation.[4,5] Briefly, in DAF, small gas bubbles sized 50–100 μm are produced from the precipitation of a gas from a solution supersaturated with that gas. This condition (supersaturation) occurs when air is dispersed through the sludge in a closed and high-pressure tank. When the sludge is removed from the tank and exposed to atmospheric pressure, the previously dissolved air leaves solution in the form of fine bubbles. While in dispersed air floatation method, relatively large gas bubbles sized 500–1000 μm are produced when gas is introduced. In vacuum floatation, supersaturation occurs when the biosolid is subjected initially at atmospheric pressure, to a vacuum of approximately 230 mm Hg in a closed tank.[5]

Although all these methods have been used in wastewater sludge treatment systems, only DAF is widely used for sludge thickening in the United States. At most wastewater treatment facilities, it was found that DAF is the most prevalent floatation thickening technology employed for treating sludge. Therefore, DAF will be the only floatation thickening technology discussed in this chapter.

3.1.3.1. Dissolved air floatation

Historically, the first municipal DAF thickener has been installed in the Bay Park Sewerage Treatment Plant, Nassau County, New York in 1957.[5] More than 700 units of DAF thickeners have been installed in the United States. Although the main function of DAF thickener is to thicken WAS, however, about 20% of the installations handle other types of sludge.

DAF thickening concentrates solids because of the attachment of microscopic air bubbles to SS. This process also helps to reduce their specific gravity to less than that of water. The attached particles then float to the surface of the thickener tank for removal by a skimming mechanism. In DAF thickening process, air is introduced to the sludge at a pressure in excess at atmospheric pressure. When the pressure is reduced to atmospheric pressure and turbulence is created, air in excess of that required for saturation leaves the solution as fine bubbles 50–100 μm in diameter. These bubbles attach to the suspended particles or trapped in the solids matrix. Since the average density of the solids–air aggregate is less than that of water, they float on the surface of solution. Normally, good solid floatation occurs with a solids–air aggregate specific gravity of 0.6–0.7. The clarified effluent (subnatant) produced is removed from the DAF thickener and returned back to the plant. In order to minimize the possibility of solid being lost in the clarified effluent, the thickened sludge (floating solids on the water surface) is removed by a skimming mechanism similar to a scum skimming system.[1] The schematic illustration of a typical DAF thickener is presented in Fig. 7.

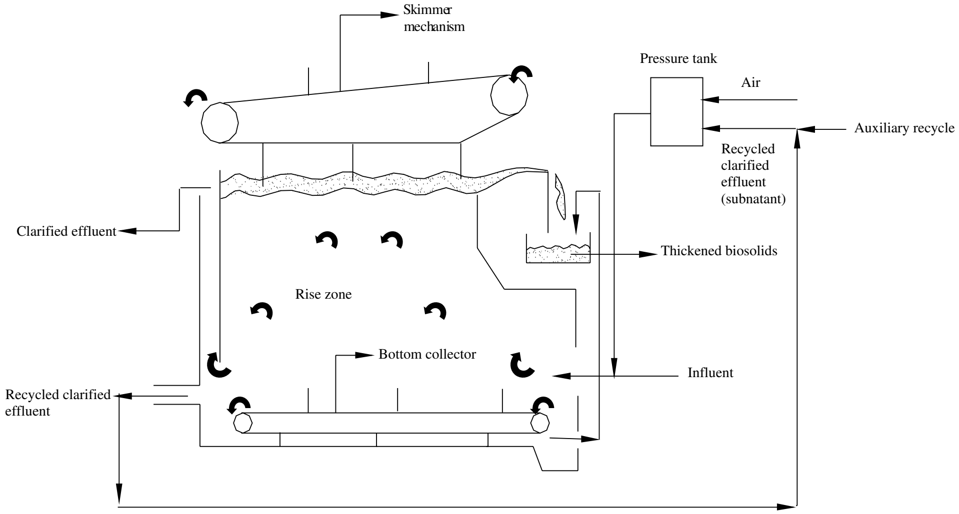


Figure 7. DAF Thickener.[24]

Polymer is frequently used to enhance DAF thickener performance. However, DAF thickener can be designed to operate with or without polymer addition. The addition of polymer during the DAF thickening process will increase the solids loading rates and solid capture. However, it is less effectively increases float solids concentrations.[27] The use of polymers as floatation aids is effective in increasing the solid recovery in the floated sludge from 85% to 98 or 99%. The dosages used for thickening WAS are 2–5 kg of dry polymer per milligram of dry solids.[4] To obtain the best result, polymer should be introduced at the point where the recycle flow and the solids feed are mixed. Table 23 presents the advantages and disadvantages of DAF thickeners compared to other major thickening equipment.

There are three different modes in which the pressurization system can be operated, which are full, partial, and recycle pressurization modes.[5] In full pressurization mode, the entire sludge flow is pumped through the pressurization tank and the air saturated sludge is then passed through a pressure reduction valve before entering the floatation tank. In this tank, the air is able to partition from the solution in the form of small bubbles. The advantage of pressurizing the full flow is it can minimize the size of floatation tank. Indirectly, it helps to reduce the capital cost.

Meanwhile, in partial pressurization mode, only a part of the sludge flow is pumped through the pressurization tank. After pressurization, the pressurized and unpressurized streams are combined and mixed before entering the floatation tank. In this arrangement, the pressurizing pump and pressure vessel are smaller. However, the size of the floatation tank would be the same as that for a full pressurization system.

The final method is known as recycle pressurization. In this method, a portion of the clarified effluent (subnatant) is saturated with air in the pressurization tank. After that, it is combined and mixed with the sludge feed before it is released into

Table 23. Advantages and Disadvantages of DAF Thickener.[5]

Advantages

- Provides better solid–liquid separation than a gravity thickener.
- Produces higher solids concentration than gravity thickener for many types of sludges.
- Requires small or less space than gravity thickener.
- Offers excellent sludge equalization control.
- Less odor problems compared to gravity thickener.
- Able to remove grit from sludge processing system.
- Able to remove fat and grease.

Disadvantages

- High operating cost compared to gravity thickener.
- The concentration of thickened sludge produced is less than in a centrifuge.
- Requires large space than a centrifuge.
- Has very little sludge storage capacity.

the floatation tank. The advantage of recycle pressurization mode when compared to other modes is that it can minimize high shear conditions. Apart from that, the recycle pressurization mode has a potential to eliminate clogging problems with pressurization pump, retention tank, and pressure release valve. Based on those advantages, the recycle pressurization mode has been chosen as the most commonly used unit in the United States.

Normally, DAF thickeners can be applied to thicken wastewater solids before dewatering or stabilization. Besides that, it can be used to thicken aerobically digested sludge or other solids before disposal or dewatering.

3.1.4. *Gravity Belt Thickening*

Recently, gravity belt thickening process has become as one of a popular methods for thickening WAS.[27] The design and operation of this process are originated from the application of belt presses in sludge dewatering. The advantages of gravity belt thickeners are: they require low energy, they are easy to operate, and they need limited attention following start up.

This equipment includes a gravity belt that moves over roller driven by a variable speed drive unit.[4] In gravity belt thickening, solids are concentrated when free water drains by gravity through a porous horizontal belt. Usually, during the operation, polymer is used for sludge conditioning before it is fed or transferred into the distribution box (feed). During this process, the sludge is distributed equally across the width of the moving belt by the distribution box. At the same time, water drains through the moving belt. It has been reported that approximately 50% of the water is lost during this process.[1] As a result, the solid content produced will be increased. When the moisture is lost, the sludge is conveyed toward the discharge end of the gravity belt thickener. Then, the sludge is arranged in the form of a long and narrow trench by a series of plow blades that is placed along the travel of the belt. This process allowing the water released from the sludge to pass through the belt. Finally, the belt travels through a wash cycle after the thickened sludge is removed. Particularly, gravity belt thickeners are suitable for the thickening of WAS before further processing. In addition, this method also applicable for thickening digested sludges for volume reduction before transportation.[27]

The performance of gravity belt thickeners for a particular solid include polymer dose, sludge feed rate, sludge characteristics, polymer and solids mixings, belt speed, belt tension, belt type, ramp use, and angle and plow configurations.[1,27]

3.1.5. *Rotary Drum Thickening*

Rotary drum thickening is also one of the systems used to thicken sludges. The function of a rotary drum thickener is similar to a gravity belt thickener where it allows the water to drain through a porous medium while solids are retained on the media. This system is often used as a prethickening step with belt filter press dewatering.[27] In the pulp and paper industries, they are suitable to be used for the thickening of high fiber sludges. Meanwhile, in wastewater treatment, this system is applicable to thicken either raw or digested sludges that contain a significant primary solids fraction.

Rotary drum thickening system consists of two important parts, which are WAS conditioning system and rotating cylindrical screens.[4] Normally, the drum is equipped with a center shaft mounted on a steel frame or is mounted on four wheels that supporting its outer perimeter. During the thickening process, polymer is added to the sludge and mixed in the conditioning drum. The use of polymer in the rotary drum thickening process for WAS, for example, can produce a thickened sludge with the range of 3–4% solids content.[1,4] Then, the conditioned sludge is passed to rotating screen drums that separate the flocculated solids from water. The thickened sludge moves out at the end of the drums while the separated water decants through the screen opening. The application of rotary drum thickener in the thickening process offers several advantages such as low energy use, less space requirements, and low capital cost. In general, the performance of a drum thickener is similar to gravity belt thickeners.

3.2. *Stabilization*

In biosolids treatment, stabilization process is one of the treatments used to stabilize the liquid sludge after the thickening process and before it can be transferred to conditioning and dewatering operations.[1] The main purposes of biosolids/sludge stabilization are listed below[2,28,29]:

i. to reduce or destroy pathogen (disease-causing organisms),
ii. to reduce volume,
iii. to minimize the potential for odor generation, and
iv. to stabilize organic matter.

Biosolids are widely used in home gardening, commercial agriculture, landscaping, recreational area, etc. They are rich in nutrients such as phosphorus and nitrogen and also contain valuable micronutrient that helps to stimulate plant growth or soil amendment to enhance certain physical properties of soil. Biosolids must be stabilized to some extent before they can be used or disposed of. The Environmental Protection Agency's (EPA) 40 Code of Federal Regulations (CFR) Part 503, *Standards for the Use and Disposal of Sewage Sludge* has stated that wastewater solids must be processed or treated before they can be used on land or as a natural fertilizer.[29] There are five major methods of stabilization that will be discussed in this section, which include alkaline stabilization, anaerobic digestion, aerobic digestion, composting, and heat drying.[30]

3.2.1. *Alkaline (Lime) Stabilization*

3.2.1.1. General description

Typically, alkaline stabilization process has been used for several years to reduce odors and to eliminate pathogen in biosolids.[31] It is a very simple process and it offers several advantages such as require low cost and simplicity of operation compared to other stabilization processes. In this treatment, lime is most commonly used for sludge stabilization.[32] However, other alkaline materials such as cement kiln dust, lime kiln dust, Portland cement, and fly ash have also been used as substitute for sludge stabilization process.[30]

Lime stabilization of sludge is one of the processes listed in the 40 CFR Part 503 Rule and it plays an important role in reducing pathogens to a level considered as Class B biosolids.[33] During this process, sufficient lime is added to the biosolids to raise the pH to 12 after 2 h of contact. Generally, there are two types of biosolids regarding to pathogen reduction that have been stated in the Part 503 Rule i.e. Class A and Class B. These classifications are based on the level of pathogens present in biosolids that are used or disposed of. The biosolids meet the Class A if pathogens such as *Salmonella* sp. and bacteria are below the detectable levels. Meanwhile, biosolids are classified as Class B if pathogens are detectable but have been reduced to levels that do not give any harm to public health and environment. However, it is found that both classes of biosolids are safe but additional requirements are necessary with Class B biosolids. The application of alkaline stabilization on biosolids can achieve the minimum requirements for both Class A and Class B biosolids with respect to pathogens.

Theoretically, the addition of lime to biosolids can help to reduce the levels of odors and pathogen by producing a high pH environment that inhibits the biological activity. During the anaerobic decomposition of organic matter, gases containing nitrogen and sulfur are evolved. These gases are the main source of odors in biosolids.[34] During the lime stabilization process, lime is added to the biosolids to increase its pH to 12.0 or higher. This pH should be remained for at least 2 h.[4,29,30] As a result, the elevated pH helps to destroy and inactivate the pathogens and microorganisms in the biosolids with highly alkaline environment. In addition, high pH can also retard the microbial activities, which can lead to odor production and vector attraction.

Figure 8 depicts the conceptual design for the lime stabilization system. Most of all sludge is passed through the grinder before entering the stabilization system. This step will help to improve the sludge mixing and flow characteristics. In addition,

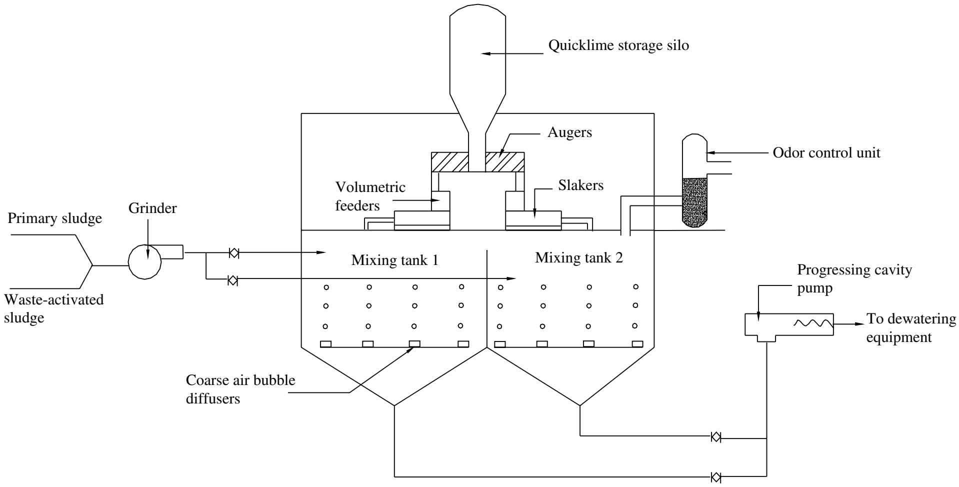


Figure 8. Conceptual Design for a Lime Stabilization System.[5]

it also protects downstream processing equipment such as pump and dewatering equipment. There are two mixing tanks provided in the system. Each tank has the capacity to treat the total sludge produced in 8 h shift. The sludge in the second tank is dosed with lime and mixed for 30 min to get pH 12.5 while the sludge is filled in the first tank at the same time. After the liming process has finished, the stabilized sludge then conveyed to the dewatering equipment. Besides that, this system also equipped with the coarse air bubble diffuser to mix the sludge with the lime slurry. In order to remove odors during the process, odor-control unit has been attached to the system.

3.2.1.2. Types of lime

Many years ago, there are two types of lime that are commercially used in biosolids stabilization, i.e. quicklime (CaO) and hydrated lime [$Ca(OH)_2$].[5,30] These alkaline materials are added to either liquid biosolids before dewatering process or dewatered biosolids in a mixer. Lime is a caustic material that can burn or damage organic tissue by chemical action. For safety handling, special equipment must be designed and it should include eyewash fountains, safety showers, protective clothing, and complete standard operating procedures. Typically, quicklime is the product resulting from the calcination of limestone by a high temperature. It consists of the oxides of calcium and magnesium. Quicklime can be classified into three major classes that are[35,36]:

i. high calcium quicklime — containing less than 5% MgO and 85–90% CaO,
ii. magnesium quicklime — containing 5–35% MgO and 60–90% CaO, and
iii. dolomite quicklime — containing 35–40% MgO and 55–60% CaO.

In wastewater treatment, high calcium quicklime, which containing 85–90% of CaO, is the most commonly used in lime stabilization process. Meanwhile, hydrated lime [Ca $(OH)_2$] is obtained by treating quicklime with water under exothermic reaction. This process is known as slaking. The reaction is described in Eq. (6):

$$\underset{\text{(Quicklime)}}{CaO} + H_2O \longrightarrow \underset{\text{(Hydrated lime)}}{Ca(OH)_2} + \text{Heat} \tag{6}$$

During the slaking process, coarse particles (CaO) are crushed and splitting them into smaller particles of $Ca(OH)_2$. These smaller particles are highly reactive and they have a large total surface area. Table 24 lists the characteristics of quicklime (CaO) and hydrated lime [Ca $(OH)_2$].

3.2.1.3. Factors affecting lime stabilization process

In lime stabilization system, there are three fundamental design parameters that must be considered, which are pH, contact time, and lime dosage. Regarding to pH and contact time, the design objective is to maintain pH greater than 12.0 and should be remained for 2 h. This action should be considered to ensure pathogen destruction and to provide high alkalinity environment. So that, the pH does not drop below 11.0 for few days. In order to accomplish these objectives, the recommended three design criteria are:

i. sludge should be treated while it is still in the liquid state,
ii. adjust the sludge pH to 12.5 by adding lime, and
iii. pH above 12.5 should be maintained for 30 min.

Basically, the required amount of lime dosage for stabilization can be determined by the following aspects such as type of sludge to be treated, chemical composition, and solids concentrations. Typically, primary sludges needs less lime dosages while the highest dosages were required to increase the pH level of WAS. In addition, iron and alum sludges also required high lime dosages. Table 25 shows the results of plant-scale tests at Lebanon, Ohio. During this test, it was found that the addition of lime ranging from 6% to 51% of the total dry solids in the sludges was required to elevate the pH level. These lime dosages were considered sufficient to maintain the sludge pH above 12.0 for 30 min.

Generally, if the lime stabilization is properly managed, significant pathogen destructions can be achieved in sludges that have been treated with lime at pH 12.0.[34,36] It was reported that in the full-scale project at Lebanon, Ohio, the indicator organisms that were used during lime stabilization process were *Salmonella*, *Pseudomonas aeruginosa*, total coliforms, fecal coliforms, and fecal streptococci. The pathogen concentrations for the raw sludges (before lime stabilization) and lime-stabilized sludges (after lime stabilization) at Lebanon, Ohio are shown in

Table 24. Characteristics of Quicklime and Hydrated Lime.[5]

Common Name/Formula	Available Form	Containers and Requirements	Appearance and Properties	Weight, lb/cu ft (Bulk Density)	Commercial Strength	Solubility in Water
Quicklime/CaO	Pebble Crushed Lump Ground Pulverized	80–100 lb moisture proof bags, wooden barrels and carloads. Store dry/maximum 60 days in tight container or three months in moisture proof bag	White (light gray, tan) lumps to powder. Unstable, caustic irritant. Slakes to hydroxide slurry evolving heat (490 Btu/lb). Air slakes to $CaCO_3$. Saturated solution approximately pH 12.5	55–75, to calculate hopper capacity-use 55; specific gravity 3.2–3.4	70–96% CaO (Below 88% can be poor quality)	Reacts to form $Ca(OH)_2$. Each lb of quicklime will form 1.16 to 1.32 lb of $Ca(OH)_2$, With 2–12% grit, depending on purity
Hydrated lime/ $Ca(OH)_2$	Powder (passes 200 mesh)	50 lb bags, 100 lb barrels and carloads. Store dry; maximum 1 year	White, 200–400 mesh, powder free of lumps; caustic dusty irritant; absorbs H_2O and CO_2 from air to form $Ca(HCO_3)_2$. Saturated solution approximately pH 12.4	25–40, to calculate hopper capacity — use 30; specific gravity 2.3–2.4	$Ca(OH)_2$ — 82–98% CaO −62–74% (Standard 70%)	10 lb/1,000 gallons at 70°F 5.6 lb/1,000 gallons at 175°F

Notes: 1 lb = 0.454 kg; 100 Btu/lb = 55 kg-cal/kg; 1 lb/cu ft = 16 kg/m^3; and 1 b/1000 gal = 0.120 g/L.

Table 25. Lime Requirement to Attain pH 12 for 30 min at Lebanon, Ohio.[34]

	Solids Concentration, %		Lime Dosage, lb Ca $(OH)_2$		pH, Average	
Sludge Type	Range	Average	Range	Average	Initial	Final
Primary sludge[a]	3–6	4.3	0.06–0.17	0.12	6.7	12.7
WAS	1–1.5	1.3	0.21–0.43	0.30	7.1	12.6
Anaerobically digested mixed sludge	6–7	5.5	0.14–0.25	0.19	7.2	12.4
Septage	1–4.5	2.7	0.09–0.51	0.20	7.3	12.7

[a] Include some portion of WAS.

Tables 26 and 27, respectively. From this study, it shows that the concentration of *Salmonella* and *Pseudomonas aeruginosa* was reduced below the level of detection (near to zero) in all types of sludges. After the stabilization process, the concentrations of fecal and total coliforms were reduced more than 99.99% in the primary and septic sludges. While in WAS, the concentrations of total and fecal coliforms were decreased to 99.97% and 99.94%, respectively. In addition, it was found that the percentage of fecal streptococci kills in primary sludge, WAS, septic sludge, and anaerobically digested sludge are 99.93%, 99.41%, 99.90%, and 96.81% respectively.[35]

3.2.1.4. Advantages and disadvantages of lime stabilization

The lime-stabilized sludge produced from the stabilization process has been used widely in many activities such as for landscaping, agriculture, mine reclamation, or as a daily landfill cover.[29] Lime stabilization offers several advantages in sludge treatment. Despite the well-known advantages, there are some disadvantages compared to other process and these have been tabulated in Table 28.

3.2.2. *Anaerobic Digestion*

3.2.2.1. General description

AD is among the oldest and most important process for the stabilization of sludge before final disposal.[5,28] It has been employed worldwide a few decades ago in most of the WWTPs. AD can be defined as a biological process that occurs naturally when various groups of bacteria and microorganisms break down organic material in the environments with the absence of oxygen. Almost any organic material can be processed with AD. These include sewage sludge from the WWTP, grass clippings, food waste, industrial effluents, animal waste, and also waste paper and cardboard. In the municipal and industrial wastewater treatments, it was found that AD plays a major role for the stabilization of concentrated sludge produced during the treatment

Table 26. Pathogen Concentration for Raw Sludges at Lebanon, Ohio (Before Lime Stabilization).[35]

Parameter	Raw Primary Sludge	WAS	Anaerobically Digested Sludge	Septage Sludge
Salmonella avg., number/100 mL	62	6	6	6
Salmonella range., number/100 mL	11–240	3–9	3–30	3–9
Pseudomonas aeruginosa, avg. number/100 mL	195	5.5×10^3	42	754
Pseudomonas aeruginosa, range number/100 mL	75–440	$91 - 1.1 \times 10^4$	3–240	$14 - 2.1 \times 10^3$
Fecal coliform avg., MF number/100 mL	N/A	2.65×10^7	2.6×10^5	1.5×10^7
Fecal coliform range, MF number/100 mL	N/A	$2.0 \times 10^7 - 3.3 \times 10^7$	$3.4 \times 10^4 - 6.5 \times 10^5$	$1.0 \times 10^7 - 1.8 \times 10^7$
Fecal coliform avg., MPN number/100 mL	8.3×10^8	N/A	1.45×10^6	N/A
Fecal coliform range, MPN number/100 mL	$1.3 \times 10^8 - 3.3 \times 10^9$	N/A	$1.9 \times 10^5 - 4.9 \times 10^6$	N/A
Total coliform avg., MF number/100 mL	N/A	8.33×10^8	2.42×10^7	2.89×10^8
Total coliform range, MF number/100 mL	N/A	$1.66 \times 10^8 - 1.5 \times 10^9$	$1.3 \times 10^5 - 1.8 \times 10^8$	$1.8 \times 10^7 - 7 \times 10^8$
Total coliform avg., MPN number/100 mL	2.9×10^9	N/A	2.78×10^7	N/A
Total coliform range, MPN number/100 mL	$1.3 \times 10^9 - 3.5 \times 10^9$	N/A		N/A
Fecal streptococci avg., number/100 mL	3.9×10^7	1.03×10^7	2.7×10^5	6.7×10^5
Fecal streptococci range, number/100 mL	$2.6 \times 10^7 - 5.2 \times 10^7$	$5.0 \times 10^5 - 2.0 \times 10^7$		$3.3 \times 10^5 - 1.2 \times 10^6$

process.[4] Furthermore, this process has become as one of the most widely used for sludge stabilizations because of its methane recovery potential.[2,30]

3.2.2.2. AD process/stages

AD is a multi-stage process. Basically, AD occurs in four stages, i.e. hydrolysis, acidogenesis, acetogenesis, and methanogenesis.[37] These stages are depicted in Fig. 9. The first stage of AD is hydrolysis. This stage also known as liquefaction. During this stage, the fermentative microorganism secretes enzymes and converts the insoluble organic materials from raw sludge such as carbohydrates, fats, and proteins into soluble organic substances such as sugars, fatty acids, and amino acids. During the hydrolysis stage, the complex organic matter is hydrolyzed by hydrolytic enzymes

Table 27. Pathogen Concentration for Raw Sludges at Lebanon, Ohio (After Lime Stabilization).[35]

Parameter	Raw Primary Sludge	WAS	Anaerobically Digested Sludge	Septage Sludge
Salmonella avg., number/100 mL	<3*	<3*	<3*	<3*
Salmonella range., number/100 mL	<3*	<3*	<3*	<3*
Pseudomonas aeruginosa, avg. number/100 mL	<3*	13	<3*	<3*
Pseudomonas aeruginosa, range number/100 mL	<3*	<3* – 26	<3*	<3*
Fecal coliform avg., MF number/100 mL	N/A	1.62×10^4	3.3×10^3	2.65×10^2
Fecal coliform range, MF number/100 mL	N/A	3.3×10^2 – 3.2×10^4	3.3×10^3	2.0×10^2 – 3.3×10^2
Fecal coliform avg., MPN number/100 mL	5.93×10^3	N/A	18	N/A
Fecal coliform range, MPN number/100 mL	560 – 1.7×10^4	N/A	18	N/A
Total coliform avg., MF number/100 mL	N/A	2.12×10^5	N/A	2.1×10^3
Total coliform range, MF number/100 mL	N/A	3.3×10^3 – 4.2×10^5	N/A	200 – 4×10^3
Total coliform avg., MPN number/100 mL	1.15×10^5	N/A	18	N/A
Total coliform range, MPN number/100 mL	640 – 5.4×10^5	N/A	18	N/A
Fecal streptococci avg., number/100 mL	1.62×10^4	6.75×10^3	8.6×10^3	665
Fecal streptococci range, number/100 mL	4.0×10^3 – 5.5×10^4	1.5×10^3 – 1.35×10^4	3.3×10^2 – 1.4×10^4	3.3×10^2 – 1.0×10^3

such as lipase, protease, cellulase, and amylase that are secreted by microorganisms. For example, the complex organic matters such as proteins are hydrolyzed to amino acids or peptides by protease enzyme. Subsequently, in the second stage, the products (sugars, fatty acids, and amino acids) formed during the hydrolysis stage are further digested during acetogenesis stage. During this phase, the fermentative bacteria or also known as acedogenic bacteria play significant role to breakdown all the soluble organic substances to produce volatile fatty acids along with other products such as hydrogen (H_2), carbon dioxide (CO_2), and ammonia (NH_3). Acetogenesis is the third stage of AD. During this stage, acetogenic bacteria convert the higher organic

Table 28. The Advantages and Disadvantages of Lime Stabilization Process[29].

Advantages
• Requires small or less space. • Flexible operation, easily started and stopped. • Simple technology. • Produce a suitable product for a variety of uses and able to be sold.
Disadvantages
• Product obtained is not suitable to be used on all type of soil. • Require higher transportation cost. • Potential of odor generation during the processing. • Potential for dust production. • Lower nitrogen content in the final product. • Potential for pathogen regrowth if the pH drops below 9.5 during storage.

acids and alcohols into simple organic acids such as acetic acids (CH_3COOH), propionic acid (CH_3CH_2COOH), and butyric acid (CH_3CH_2 CH_2COOH). These acetic acids are produced along with hydrogen (H_2) and carbon dioxide (CO_2). Finally, methane (CH_4) and carbon dioxide (CO_2) are produced in the methanogenesis stage. At this stage, the bacteria called methane formers or also known as methanogens digest the intermediate products from the previous stages (hydrolysis, acidogenesis, and acetogenesis) and transform them into methane (CH_4), carbon dioxide (CO_2), and water. The methane gas produced can be burnt to generate heat or electricity for plant operation or even used as vehicle fuel. In addition to methane (biogas), AD also produces a solid and liquid by-product called digestate. This nutrient-rich by-product can be used as soil conditioner or as a natural fertilizer after some treatment.

3.2.2.3. Factors affecting AD process

In the anaerobic environment, there are various important parameters that affect the rates of digestion process. These include[4]:

i. temperature,
ii. pH,
iii. solid retention time (SRT),
iv. hydraulic retention time (HRT),
v. presence of toxic materials, and
vi. bioavailability of nutrients and trace metals.

However, it was found that the first four parameters are the most significant factors and will be discussed in this chapter.

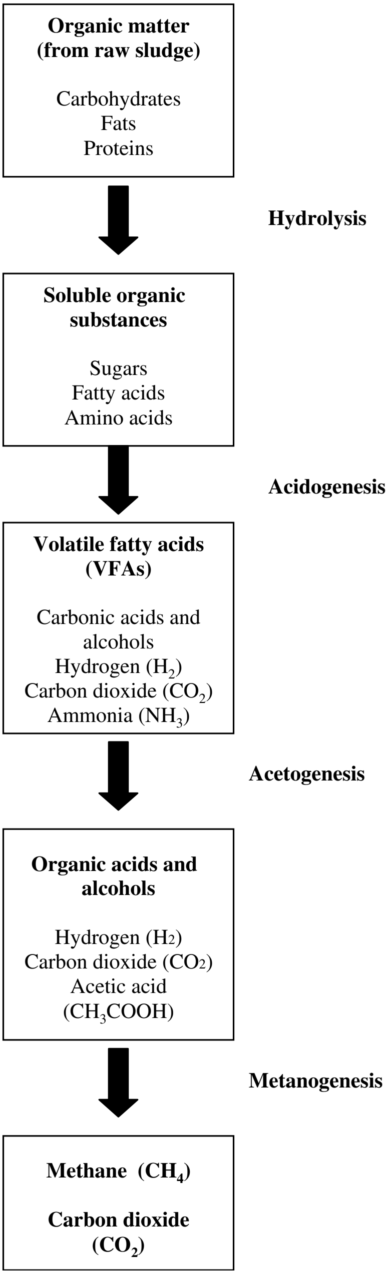


Figure 9. Four Stages in AD.

3.2.2.3.1. *Temperature*

Temperature is considered as the most important factor in the successful of AD process.[35,39] Generally, the increasing of temperature can affect the metabolism and growth rate of microorganisms. There are mainly two temperature ranges that provide optimum digestion conditions for the production of methane which include mesophilic and thermophilic ranges. The mesophilic range is between 20°C and 40°C while the thermophilic range is between 50°C and 65°C. Several studies have been reported that the optimum temperatures for the AD process are 35°C (mesophilic) and 55°C (thermophilic), respectively.[40–41] The digestion process that occurs at high temperature (thermophilic conditions) offers several benefits compared to mesophilic conditions, which include[37,40,42]:

- increase the rates of destruction of pathogens and weed seeds,
- accelerate biological and chemical reaction rates with increasing temperature,
- increase the solubility of the organic compounds, and
- enhance higher biogas production.

However, there are some disadvantages of thermophilic conditions, when compared to mesophilic conditions, which include[35,37,40,43,44]:

- require higher energy for heating,
- the process is unstable due to higher temperature,
- produce poor or low quality effluent,
- sensitive to environmental changes, and
- increase the fraction of free ammonia that inhibits the role of microorganisms to breakdown the organic materials.

3.2.2.3.2. *pH*

The parameter pH is the second leading parameter that affects AD process. In addition to temperature, pH is also one of the significant factors that affect the bacterial growth. It was found that each group of microorganisms or bacteria has a different optimum pH range. Anaerobic bacteria especially methanogenic bacteria (methanogen) are sensitive to the acid concentration in the anaerobic digester. The presence of acidic environment can inhibit the bacterial growth. Methanogenic bacteria require a neutral environment to slightly alkaline environment in order to produce methane. Several studies have reported that various optimum pH range for the production of methane are between 6.8–7.2,[39] 6.5–7.2,[37]and 7.0–7.2.[45] Meanwhile, acid-forming bacteria or fermentative bacteria are less sensitive to acidic environment. Thus, they can function very well in the pH range between 4.0 and 8.5 during the process.[37]

3.2.2.3.3. *Retention time — solids retention time and hydraulic retention time*

Another parameter that affects the AD process is retention time, which includes SRT and HRT. Most AD systems are designed to retain the sludge for a couple of days in the digester. SRT can be defined as the average time the solids held in the digester.[37] It is well known that this parameter is the most important factor in controlling the conversion of sludge to gases such as methane and carbon dioxide. It also plays a significant role in order to maintain the stability of the anaerobic digester. Basically, the increase or decrease of SRT will lead to an increase and decrease of the rate of the reactions.[4,37] Meanwhile the average time that the liquid sludge is being retained in the digester is known as HRT. The HRT is important since it establishes the quantity of time available for bacterial growth and subsequent conversion of organic material in the sludge into gas. For the system to be economical, the AD process should occur at short HRT.[46] Generally, the required retention time for the completion of AD reaction would vary depends on the technology used, temperature range, and the composition of wastewater.

3.2.2.4. Types of AD system

Basically, there are four types of AD systems that are available to stabilize wastewater sludges. They are[35]:

i. low-rate AD,
ii. high-rate AD (single-stage and two-stage),
iii. anaerobic contact, and
iv. phase separation.

3.2.2.4.1. *Low-rate AD*

In the AD process, low-rate (standard) digestion is found to be the simplest and oldest type of sludge stabilization method. The retention time required for low-rate digestion is between 30 and 60 days.[47] Meanwhile, organic loading rates vary between 0.4 and 1.6 kg total VSS per m^3 of digester per day. Figure 10 depicts the schematic diagram of low-rate (standard) digestion. Normally, large storage tank is used as a low-rate digester. During the process, sludge is fed into the digester intermittently. In the digester, the sludge is not heated or mixed. After the sludge is fed in the digester, bubbles of gas are generated and they rise to the surface. The rising gas to the surface provides the only source of mixing in the digester.[48] The rising gases also carry the lighter sludge particles to the surface to form scum. Basically, the stratification occurs in four zones, which include scum layer, supernatant (liquid layer), the layer of actively digesting sludge, and the layer of digested or stabilized

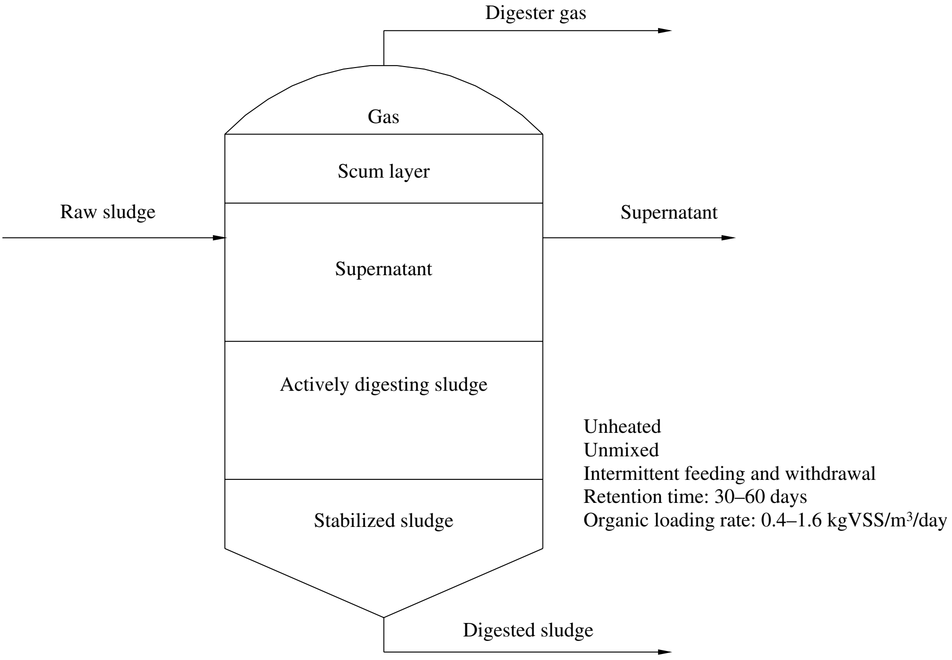


Figure 10. Schematic diagram of low-rate AD system.[5]

sludge. The stabilized or digested sludge formed at the bottom of the digester is periodically removed from the center of the floor. Meanwhile, the supernatant is withdrawn from the side of the digester and recycled back to the WWTP. At the same time, the biogas produced from the digestion process is drawn off through the digester cover. The low-rate digestion process offers a major disadvantage where the large tank volume is needed because of long retention times, low loading rates and the formation of thick scum layer.[47]

3.2.2.4.2. *High-rate AD (single-stage and two-stage)*

High-rate AD is the most commonly used system in the United State.[35] This digester has experienced a major improvement from the low-rate (standard) digestion system since 1950s. The high-rate digestion system has two types of digesters, which are single-stage and two-stage high-rate digesters. There are four elements of high-rate digestion that should be considered in order to create a steady and uniform environment and the best conditions for the biological process. These include the value of heating, auxiliary mixing, thickening the raw sludge, and uniform feeding. In the single-stage digester, the sludge is mixed thoroughly to create homogenous environment throughout the digester and completely heated to increase the rate of digestion process. As a result, the distribution of temperature is more uniform throughout the digester because the sludge is thoroughly mixed. In this process, heating is important in order to increase the rate of microbial growth and, therefore, increase the rate of digestion with increasing temperature. Normally, high-rate digester can operate at two temperature ranges, i.e. mesophilic range and thermophilic range (this range has been discussed above). Most commonly, the sludge in the digester is heated by external heat exchangers. The reason for using external heat exchangers is because of their flexibility and ease of maintenance.[37] However, other heating methods such as internal heat exchanger, steam injection, and direct flame heating also can be used to heat sludge in the digester during the process.[1] Uniform sludge feeding is very important in the high-rate AD. The sludge is fed continuously to help maintain steady-state conditions in the digester. Thickening the sludge before the digestion process increased the SRT and thereby enhanced solids reduction. The retention time required for this stage is between 10 and 15 days, while organic loading rate vary between 1.6 and 8.0 kg VSS/m^3/day.[47] Single-stage high-rate digestion system is illustrated in Fig. 11.

Two-stage AD has been considered more effective than the conventional single-stage system in the conversion of the sludge to biogas. As shown in Fig. 12, high-rate digester is coupled in series with a second tank. The first tank is known as primary digester while the second tank is secondary digester. The first stage (primary digester) is the active stage. The important characteristics of this stage are: the digesting sludge is thoroughly mixed and the retention time only takes a few days. Traditionally, the second digester is similar in design to the primary digester except that it is neither heated nor mixed. After the digestion process, effluent produced from primary digester is discharged into the secondary digester. The main function of the

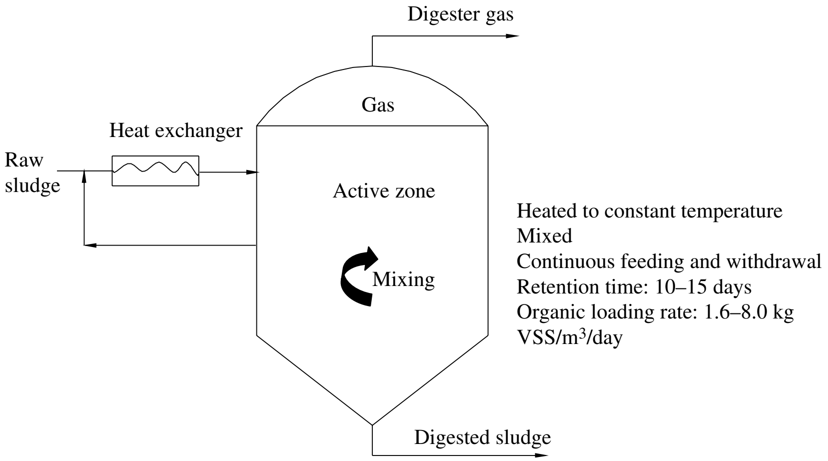


Figure 11. Schematic Diagram of Single-Stage High-Rate Digestion System.[5]

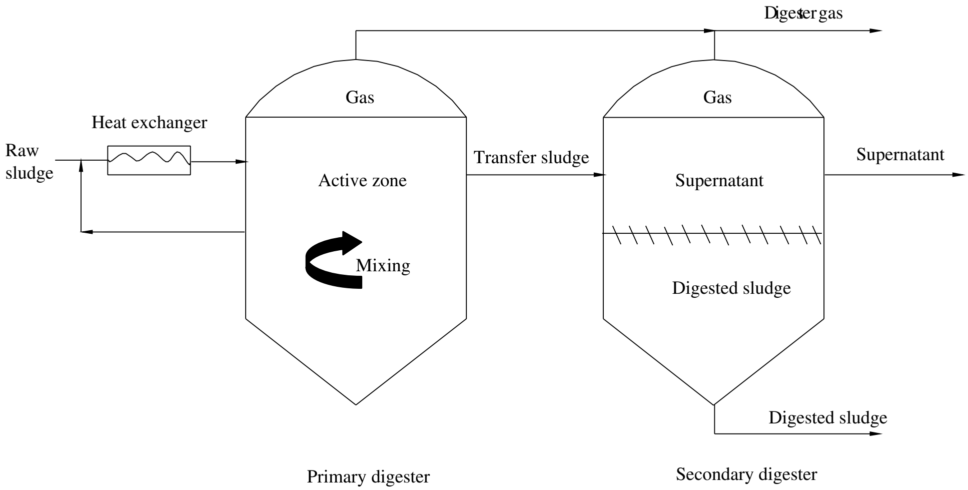


Figure 12. Schematic Diagram of Two-Stage High-Rate Digestion System.[5]

secondary digester is to allow gravity thickening of digested sludge and decanting the supernatant liquor. The second digester also can serve as a settling chamber in which the digester contents separate into two layers, namely digested sludge and supernatant. Both tanks may be equipped with fixed or floating covers. Usually, second-stage digester covers used floating type, where it can also be employed to store the digester gas produced from the digestion process. Unfortunately, many secondary digesters have performed poorly as thickeners. As a result, they produce a dilute sludge together with a supernatant contains with high concentration of SS. The basic cause of this problem (poor performance) is that anaerobically digested sludge does not settle readily. These poor settling phenomena are commonly associated with the following two reasons[5]:

i. Floatation of solids due to the attachment of gas bubbles and sludge particles.
ii. High proportion of fine sized particles.

However, the secondary digester can function very well as following despite their poor performance in thickening anaerobically digested sludge, which include:

- Thickening digested primary sludge,
- Providing standby digester capacity (if the digester is equipped with heating, mixing, and intake piping),
- Storing digested sludge (if fitted with floating cover), and
- Ensuring against short circuiting of raw sludges through digestion (important for odor control if digested sludge is transferred to open basins or lagoons).

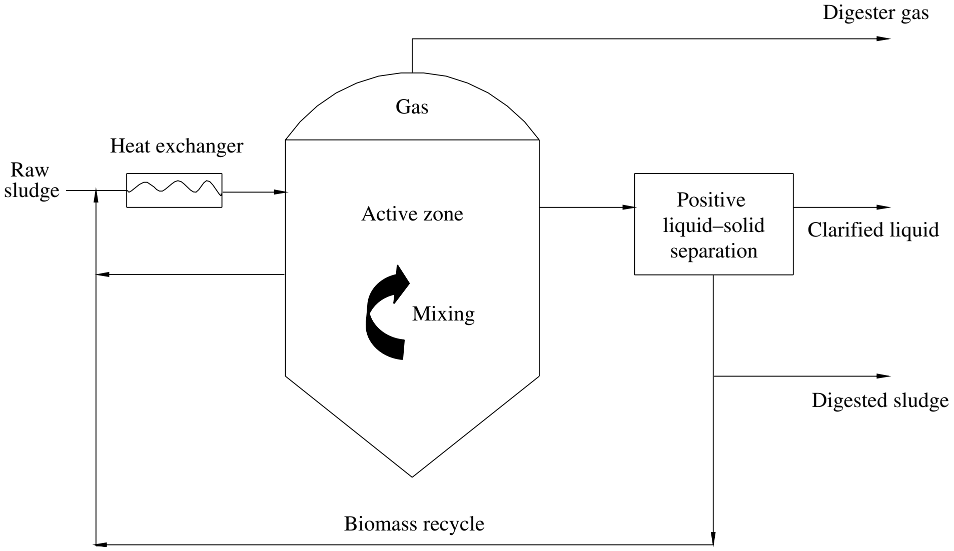


Figure 13. Schematic Diagram of Anaerobic Contact Process.[5]

3.2.2.4.3. *Anaerobic contact process*

The anaerobic contact process is a suspended growth process. This process is similar in design to the activated sludge process. The essential feature of this system is that positive liquid–solids separation through the use of a bacterial biomass is utilized. A portion of this biomass that leaving the digester is recycled back and then mixed with the incoming raw sludge as shown in Fig. 13. This recycling of sludge allows for adequate cell retention to meet kinetic requirements.[5] To obtain any of the benefits from recycling, the return stream must be more concentrated than the contents of the digester. The process usually has a vacuum degasifier placed following the anaerobic digester to eliminate gas bubbles and thereby improving cell settling.[49] The anaerobic contact process has been applied in treating high-strength industrial wastes. Meanwhile, this process has also been operated successfully at a laboratory scale to stabilize primary sludge.[50] However, this system is rarely considered in municipal anaerobic sludge digestion.

3.2.2.4.4. *Phase separation*

Basically, phase separation is also referred to multi-stage AD system. This system involves two general phases, which include acid formation and methane production. The acid-forming phases, which involve hydrolysis and volatile acid fermentation, are separated from the gas-forming phase (methane formation) by being conducted in the separate digester.[51] In the previous three AD processes (low rate, high rate, and

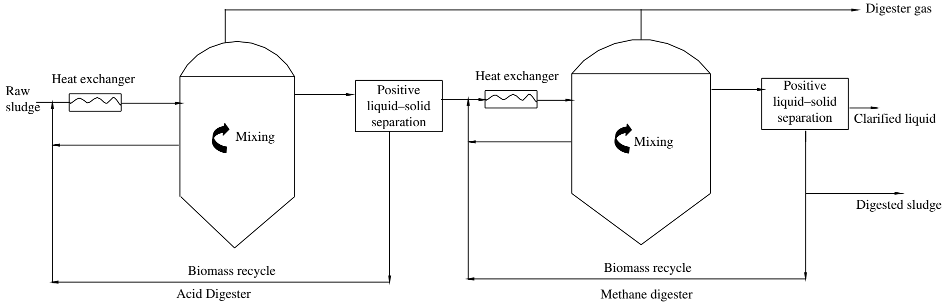


Figure 14. Schematic Diagram of Two-Phase AD Process.[5]

contact process), both phases occur in one digester. As early as 1958, the potential benefits of separating the two phases into separate tanks were discussed. Generally, it was found that two-phase digestion is feasible for the treatment of sewage sludge.[52] The first phase is known as acid-phase digester. It consists of hydrolysis and first acid production steps. During these steps, the complex organic matter is hydrolyzed and converted into soluble organic substances and volatile fatty acids. Meanwhile in the second phase, the organic material is converted further to acetic acid through acetogenesis as well as methane-formation step in which methanogenic bacteria break down the organic matter into biogas.[51] The schematic diagram of two-phase AD process is illustrated in Fig. 14. The phase separation process offers several potential advantages when compared to other processes. These are[35]:

- Capability of maintaining the optimum environment for each group of microorganisms involved in the digestion process.
- Substantial reduction in total digester volume and the consequent savings in capital and operating costs.
- Higher rates of solid stabilization and increased the production rate of biogas at the end of the process.
- Low heat requirement.
- Suitable for incorporation into existing treatment plants with minimum capital investment.

3.2.2.5. Advantages and disadvantages of AD

Overall, AD has been and continues to be as one of the most widely used processes for the stabilization of sludge produced from the WWTP. The widespread use of this process over other stabilization processes is mainly related to the potential advantages as listed in Table 29. Despite the well-known advantages of AD process, there are several disadvantages when compared to other stabilization process. The disadvantages of this process are also listed in Table 29.

Table 29. The Advantages and Disadvantages of AD Process.[46]

Advantages

- Produces methane, which is a usable source of energy.
- Reduces total sludge mass through the conversion of organic materials in methane, carbon dioxide, and water.
- End product can be used as a soil conditioner.
- Inactivates pathogens.
- Less energy required.
- Less biological sludge produced.
- Process more effectively provides sanitization/removal of diseases.

Disadvantages

- Has a high capital cost.
- Produces a poor quality sidestream.
- Slow growth rate of anaerobic organisms.
- Not suitable for treating low concentrated wastewaters.
- A relatively high operating temperature has been required for efficient performance.
- Increase potential for production of odor and corrosive gases.
- Much more sensitive to the adverse effect of lower temperature on reaction rates.
- The digestate (semi-solid) produced at the end of the process may require further treatment with an aerobic treatment process to meet discharge requirements.

3.2.3. *Aerobic Digestion*

3.2.3.1. General description

Aerobic digestion of wastewater sludges is defined as a stabilization process, in which aerobic bacteria consume the biological degradable organic component of the sludge and convert them into carbon dioxide, water, and ammonia.[30,53] The ammonia is further converted to nitrates, as the digestion process proceeds. The basic objectives of this treatment are to produce a biologically stable end product while reducing both sludges mass and volume.

Historically, the studies on aerobic digestion on wastewater sludges have been conducted since the early 1950s in the United States. It has been indicated that aerobic digestion performed as well as AD in reducing volatile solids. By 1963, at least one major equipment supplier had approximately 130 installations in plants with flow from 37.8 to 378 m^3/day. By the late 1960s and the early 1970s, aerobic digestion facilities have been suggested to be built in a new constructed treatment plants. As a result, by the early 1979, numerous plants use aerobic digestion as one of the treatment units to stabilize the sludge produced.[5] Typically, aerobic digestion is used in smaller WWTPs.[30,53] The following types of sludges have been successfully treated by aerobic digestion, which include WAS only, mixtures of WAS or trickling filter sludge and primary sludge, and waste sludge from extended aeration.[4]

However, the most widespread use of aerobic digestion has been in the treatment of WAS.

The aerobic digestion of sludge is a continuation of the activated sludge process under endogenous condition. Basically, the aerobic digestion consists of two steps involving the direct oxidation of biodegradable materials and endogenous respiration where cellular material is oxidized.[5,54,55] These steps can be illustrated as in Eqs. (7) and (8).

$$\text{Organic matter} + NH_4^+ + O_2 \longrightarrow \text{cellular material} + CO_2 + H_2O \tag{7}$$

$$\text{Cellular material} + O_2 \longrightarrow \text{digested sludge} + CO_2 + H_2O + NO_3^- \tag{8}$$

Equation (7) shows the oxidation of organic matter to cellular material. During this process, microorganisms will oxidize the organic matter in the presence of oxygen and converting them into cellular materials, carbon dioxide, and water. Meanwhile, Eq. (8) describes that the cellular material is subsequently oxidized the cellular material to digested sludge. Together with digested sludge, carbon dioxide, water, and nitrate are also produced in the second step of the oxidation. The process described in Eq. (8) is referred to as "endogenous respiration", which is the predominant reaction in aerobic digestion.[55] There are some parameters affecting the aerobic digestion process, which are[56,57]:

- temperature,
- the rate of sludge oxidation,
- system oxygen requirements,
- sludge loading rate,
- sludge age, and
- sludge solids characteristics.

3.2.3.2. Process variations of aerobic digestion

There are several process variations of aerobic digestion. These variations include conventional (mesophilic) aerobic digestion, high-purity oxygen digestion, dual digestion, and autothermal thermophilic aerobic digestion (ATAD).

3.2.3.2.1. *Conventional aerobic digestion*

Conventional aerobic digestion is also known as mesophilic aerobic digestion.[58] This variation is quite simple and plays an important role to stabilize excess activated sludge in unheated open digesters through diffused air or surface mechanical aeration. Typically, the digestion process occurs at a mesophilic temperature range. The facilities and equipment are similar to those used for activated sludge system. In a conventional aerobic digester, solids concentration must be less than 3%.[58]

Basically, sludge can be introduced to the aerobic digester on a batch, semi-batch, or continuous bases. In the batch basis, the raw sludge is filled into the digester and aerated for 2–3 weeks, then stopped. The supernatant is decanted and the settled solids are removed. For semi-batch basis, raw sludge is added every couple of days. The supernatant is decanted periodically and the settled solids are held in the digester for a long time before being removed. Usually, sludge is thickened by floatation to reduce the required digestion volume. To design an aerobic digester, a few aspects should be considered such as hydraulic detention time, organic loading, oxygen demand, power requirements, and temperature.[58] Meanwhile, the design parameters involved are hydraulic detention time, oxygen demand, mixing, and temperature.

3.2.3.2.2. *High-purity oxygen digestion*

High-purity oxygen digestion is the aerobic digestion process, in which high-purity oxygen is used instead of air as the oxygen source.[4] The sludge produced from the process is similar to sludge from conventional aerobic digestion. This process can be conducted in either open or closed tanks. In the closed tanks, high-purity oxygen atmosphere is maintained in the space above the liquid surface and oxygen is transferred into the sludge through mechanical aerators. Meanwhile, when performing the process in open tanks, oxygen is normally introduced to the sludge in small bubbles with special diffusers. The bubbles dissolve before they reach the liquid surface. The principal advantage of the system is that it is relatively insensitive to changes in ambient temperatures when closed tanks are used. This is because of the increased rate of biological activity and the exothermic nature of the process.[6] Meanwhile, the major disadvantage of this process is the increased cost associated with oxygen generation.[4] High temperatures resulting from the exothermic process increase the rate of volatile solids destruction and favoring its use in cold climate regions.

3.2.3.2.3. *Dual digestion*

Dual digestion is a two-step sludge digestion process that combines the aerobic thermophilic pretreatment prior to AD.[59] Basically, the first step of dual digestion system comprised of a high-rate autothermal aerobic digester.[60] The aerobic reactor can be maintained at thermophilic temperatures where it is greater than 45°C through the conservation of heat generated by biological oxidation of degradable organic matter in the sludge and does not require the use of an external heat source. During the process, high-purity oxygen is introduced into the aerobic digester. Using high-purity oxygen to aerate concentrated sludges greatly reduces heat loss from the system due to evaporation of water into air. The advantages of using aerobic thermophilic digestion in the first step of dual digestion system are shown in Table 30.

Table 30. The Advantages of Using Aerobic Thermophilic Digestion in the First Step of Dual Digestion System.[4]

Advantages
• It can increase levels of pathogen destruction. • It can improve volatile solids reduction. • It can increase the generation of methane gas in the second step (anaerobic digester). • Less organic material in and fewer odors produced by the stabilized sludge. • Equivalent volatile solid reductions can be achieved in one-third less tankage than a single-stage anaerobic digester.

Meanwhile, the second step of the dual digestion system consists of a high-rate anaerobic digester that receives the heated and partially digested sludge from the autothermal aerobic pretreatment step.[60] In the second step of the dual digestion system, AD is normally occurs in mesophilic temperature ranges. The anaerobic digester completes the stabilization process by further breaking down volatile matter into carbon dioxide and methane gas.

3.2.3.2.4. *Autothermal thermophilic aerobic digestion*

ATAD is an advanced sewage sludge treatment that provides hygienization. This system has been studied since the 1960s and it has been significantly developed since the mid-1970s.[61] It has been widely implemented in Germany, Great Britain, France, and Italy. Basically, ATAD represents a variation of both conventional and high-purity oxygen aerobic digestions. Generally, ATAD can degrade complex organic substances into end products including carbon dioxide and water. Some of the energy released by microbial degradation is used to form new cellular material but much of it is released as heat. In ATAD, the heat released by the digestion process is the major heat source used to achieve the desired operating temperature. Figure 15 depicts the various inputs, outputs, and heat production items to be included in a heat balance for describing aerobic thermophilic systems operation. ATAD systems are normally two-stage aerobic processes that operate under thermophilic conditions ranges from 40°C to 80°C without supplemental heat. The process has a self-heating ability and removes pathogens when operating at a temperature of at least 55°C. Typical ATAD systems operate at 55°C and can reach 60–65°C in the second stage. These systems are termed as autothermal because supplemental heat is not provided and their self-heating ability during the process. Autothermal conditions result from an adequately thickened sludge feed, a suitably insulated reactor, good mixing, and an efficient aeration device that keeps the latent heat loss to an acceptable level.[61] During the process, the feed sludge is usually thickened to provide a digester feed solids concentration of greater than 4%. The reactors are insulated to conserve the heat produced from the oxidation of volatile solids. Since the process

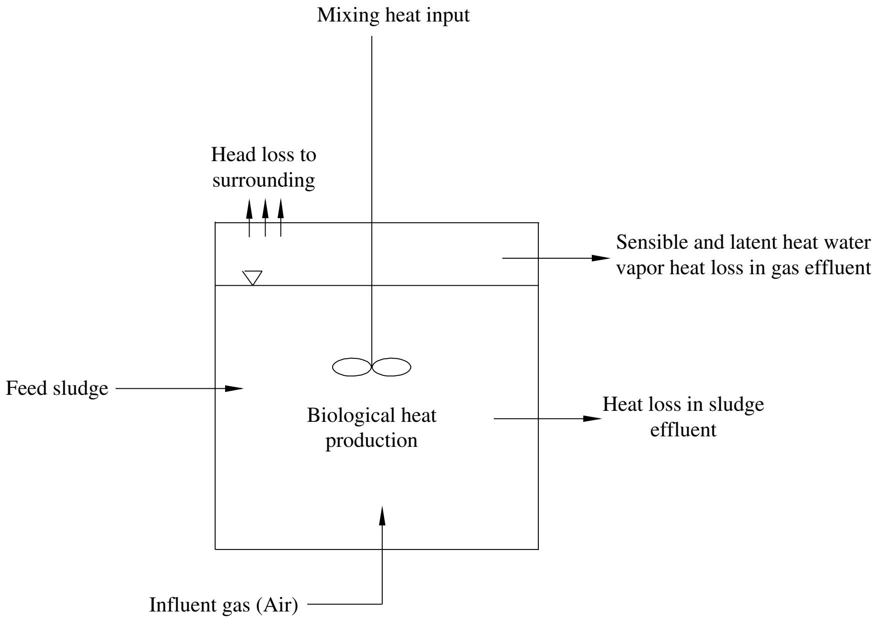


Figure 15. Heat Balance Schematic for the Autothermal Aerobic Digestion.[61]

occurs at high temperature, ATAD systems may be able to meet Class A biosolids.[62] Table 31 lists all the advantages and disadvantages claimed for this mode of operation.

3.2.3.3. Design considerations

Basically, there are several factors that should be considered when designing the aerobic digester, which include temperature, volatile solids reduction, oxygen requirements, pH reduction, mixing, and dewatering.[5]

The liquid temperatures in open tank digesters depend on weather conditions and can fluctuate extensively. As with all biological systems, lower temperatures retard the process whereas higher temperatures speed it up. In designing a digester, consideration should be given to minimize heat loss by using concrete instead of steel tanks, to place the tanks below rather than above grade, and to replace surface aeration to sub-surface aeration. Meanwhile in extremely cold climates, consideration should be given to cover the tanks, to heat the sludge, or both. Besides that, the design should allow for the necessary degree of sludge stabilization at the lowest expected liquid operating temperature and it should meet maximum oxygen requirements at the maximum expected liquid operating temperature.

Stabilizing the sludge, reducing pathogens, and reducing the mass of solids for disposal are the major objectives of aerobic digestion. The reduction in mass of

Table 31. The Advantages and Disadvantages of ATAD.[4,5,61]

Advantages

- High disinfection capability.
- Low space and need smaller reactors.
- High sludge treatment rate.
- Relatively simple technology and easy to operate.
- More economical.
- Suitable for small facilities.
- Shorter the retention times to approximately 5–6 days in order to achieve higher volatile solid reduction of 30–50%.
- Production of pasteurized sludge.
- Require less oxygen compared to mesophilic process (30–40%).
- Can produce Class A biosolids if the digesters are well mixed and maintained at 55°C where the pathogenic viruses, bacteria, and other parasites killed and reduced to below detectable levels.

Disadvantages

- It must incorporate a thickening operation.
- Poor dewatering characteristics of ATAD biosolids.
- Lack of nitrification.
- Produce objectionable odors.
- Nonoxygen aerated systems require extremely efficient aeration and insulated tanks.
- Perceived high energy requirements.

solids is possible only with the destruction of the biodegradable organic content of the sludge. Volatile solids reductions of 35–50% are attainable by aerobic digestion. The reduction in biodegradable volatile solids that occurs during the aerobic digestion can be represented by a first-order biochemical reaction as shown in Eq. (9):

$$\frac{\mathrm{dM}}{\mathrm{dt}} = -\mathrm{K_d} \tag{9}$$

where dM/dt = the rate of change of biodegradable volatile solids per unit time, $\mathrm{K_d}$ = reaction rate (time^{-1}), and M = concentration of biodegradable volatile solids remaining at time t in the aerobic digester (mass/volume).

In aerobic digesters, decreases in pH and alkalinity have been observed at increasing detention times. The drops in pH are caused by acid formation that occurs during nitrification (i.e. conversion of ammonia nitrogen to nitrate). Meanwhile, the drop in alkalinity is caused by lowering of the buffering capacity of the sludge due to air stripping. It has been observed that the system will acclimate and perform well at lower pH value. If the feed sludge has very low alkalinity and pH, the addition of chemical may have to be made to increase the alkalinity.

Adequate mixing is required in aerobic digesters to maintain solids in suspension. Solids settlement will reduce the effective volume of the digester and result in anaerobic conditions in settled solids. There are several types of aeration devices that have been used to mix the sludge in the digesters including diffused air, mechanical surface aeration, mechanical submerged turbines, draft tube aeration, jet aeration, and combined system.

Activated sludge biomass is most often represented by the empirical equation $C_5H_7NO_2$. Under the prolonged periods of aeration typical of the aerobic digestion process, Eq. (8) can be written as follows:

$$C_5H_7NO_2 + 7O_2 \longrightarrow 5CO_2 + 3H_2O + H^+ + NO_3^- \tag{10}$$

Hypothetically, this equation indicates that 0.898 kg of oxygen is required to oxidize 0.45 kg of cell mass. From pilot- and full-scale studies, however, the mass of oxygen required to degrade a 0.45 kg of volatile solids were found to be 0.789–0.939 kg. For mesophilic systems, a design value of 2.0 is recommended. For auto-thermal systems, which have temperatures above 45°C, nitrification does not occur and a value of 1.45 is recommended. Field studies have also indicated that a minimum value of 1.0 mg/L oxygen should be maintained in the digester at all times.[5]

3.2.3.4. Advantages and disadvantages of aerobic digestion

Aerobic digestion is one of the most widely used sludge stabilization processes in WWTPs. Various advantages have been claimed for aerobic digestion over other stabilization processes, particularly AD. The potential advantages of this process are listed in Table 32. Despite the well-known advantages of aerobic digestion process, there are several disadvantages when compared to other stabilization processes as listed in Table 32.

3.2.4. *Composting*

3.2.4.1. General description

Composting is a feasible and beneficial option in biosolids management. Composting is becoming a more acceptable alternative for the treatment of sewage sludge in many municipalities due to its potential use of the composting product for land application. It is one of the most promising options to transform the organic waste such as sewage sludge into a value-added product, which is known as compost, and a proven method for pathogen reduction. Composting can be defined as a biological decomposition and stabilization of organic matters, under thermophilic temperature conditions (>55°C) to produce a final product that is stable and free of pathogens and plant seeds and can be beneficially applied to land[64]. Generally, the composting process has been practiced several decades ago in wastewater sludge treatment to stabilize the organic matter prior to their use as a soil amendment or mulch in landscaping, horticulture, and agriculture, to reduce volume, and to eliminate pathogenic organisms.[28]

Table 32. The Advantages and Disadvantages of Aerobic Digestion Process.[5,63]

Advantages

- Have low capital costs for plant under 5 MGD (220 L/s) compared with AD and other processes.
- The process is relatively easy to operate compared to AD. As a result, less skilled labor can be used to operate the digester.
- The production of nuisance odors is minimal.
- Has lower levels of BOD, phosphorus, SS, and ammonia nitrogen in the supernatant.
- Fewer effects from upsets such as the presence of toxic interferences or changes in loading and pH.
- Greater reduction in grease and hexane solubles.
- Shorter retention periods.
- Greater sludge fertilizer value.
- An effective alternative for small WWTPs.
- Produce a humus-like, odorless, and biologically stable end product.
- Autothermal thermophilic digestion results in 100% pathogen destruction.
- Pathogen reductions are high under normal design.

Disadvantages

- Produce an aerobically digested sludge with very poor mechanical dewatering characteristics.
- Higher operating costs, especially energy costs.
- Performance is influenced by temperature, location, and type of tank material.
- The sludges do not dewater easily by vacuum filtration.
- Highly sensitive to ambient temperature (operation at temperatures below 15°C).
- May require excessive retention times to achieve stabilization; if heating is required, aerobic digestion may not be cost-effective).
- Less reduction in volatile solids.
- Unfavorable economics for larger WWTPs.

3.2.4.2. Types of composting methods

Basically, there are several methods used in composting; for example, windrow, aerated static pile, and in-vessel system. Each method involves mixing dewatered wastewater solids with a bulking agent to provide carbon and increase porosity. The materials that are suitable to be used as bulking agents are agricultural by-products, yard trimmings, food by-product, and industrial by-products from wood processing.[66] After mixing process, the resultant mixture is piled or placed in a vessel where microbial activity causes the temperature of the mixture to rise during an active composting period. After active composting, the material is cured and distributed.

Windrow composting is a simple method where sewage sludge and bulking agent mixture are built into large and long piles with a triangular cross-section. The cross-sectional dimensions vary with feedstock and turning equipment used. Normally, the height of the windrows is about 1.5–3.0 m and the width is about 3–6 m. Windrows can be formed with a front-end loader, dump truck, or conveyor. Periodic turning or mixing of the piles helps to aerate the pile, break up the particles to increase surface area, improve the porosity, and allow trapped heat, water vapor and gases to escape. Besides that, periodic turning is essential to move outer surfaces of material inward in order to ensure that all parts of pile are subjected to temperature of about 55°C for pathogens destruction and to ensure that adequate moisture is present throughout the compost pile.[2,65] Windrow composting process will take from several months to more than a year to reach completion. The use of windrow composting in sludge stabilization offer several advantages and disadvantages. Table 33 lists all the advantages and disadvantages of windrow composting. Meanwhile, the schematic diagram of windrow composting system is depicted in Fig. 16.

Aerated static pile composting is a more sophisticated method using an aeration system that is physically a part of the compost pile. This method is often employed when composting odorous, fine, or high moisture material such as sewage sludge. In this method, dewatered sludge is mechanically mixed with a bulking

Table 33. The Advantages and Disadvantages of Windrow Composting Method.[65,66]

Advantages

- Low capital cost.
- Proven technology on small scale.
- Adaptable to changes in sludge and bulking agent characteristics.
- Low energy requirements.
- Minimally dependent on mechanical equipment.

Disadvantages

- Large land area required.
- Need additional cost of labor for the operation of the turning equipment.
- Require large maneuvering area for the turning equipment such as front end loader.
- Highly affected by weather; but can be lessened by covering the pile.
- High potential for odor generation during turning.
- Longer composting period.

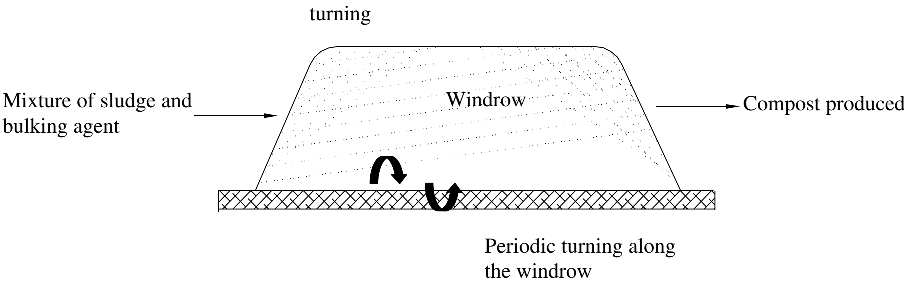


Figure 16. Schematic Diagram of Windrow Composting System.

agent attached to a blower (force ventilation) in which air is transferred to the composting material.[66] During active composting stage, the composting material is not disturbed until the composting process is nearly completed when the temperature of composting material drops to ambient temperature. After active composting, as the pile is starting to cool down, the material is moved into the curing pile for the maturation stage. Basically, the use of static pile aerated by force ventilation allows a sustained supply of oxygen and removes heat, decomposition gases, and water vapor produced during the active composting.[67] Various advantages have been claimed for aerated static pile over windrow method. The potential advantages of this method are listed in Table 34. However, aerated static pile method also has some disadvantages and these are also listed in Table 34. The schematic diagram of aerated static pile composting method is shown in Fig. 17.

In-vessel composting, also called closed composting, takes place in a completely enclosed container where the levels of temperature and oxygen can be closely monitored and controlled. A mixture of dewatered wastewater solids and bulking agent is fed into a vessel. Aeration is provided by forced air or mechanically by moving

Table 34. The Advantages and Disadvantages of Aerated Static Pile Method.[65,66,68]

Advantages

- Suitable for both small- and large-scale composting processes.
- Offer greater opportunity for odor control (a biofilter is usually constructed of a mixture of wood chips and finished compost that function to minimize odor).
- Require less land area compared with windrow composting.
- Moderate labor requirements if compared to windrow composting.
- Lower reaction time.

Disadvantages

- Highly affected by weather (can be lessened by covering with roof).
- High capital costs if compared to windrow composting.
- Large volumes of air to be treated for odor control.
- Requires a higher level of technical management than windrow composting.

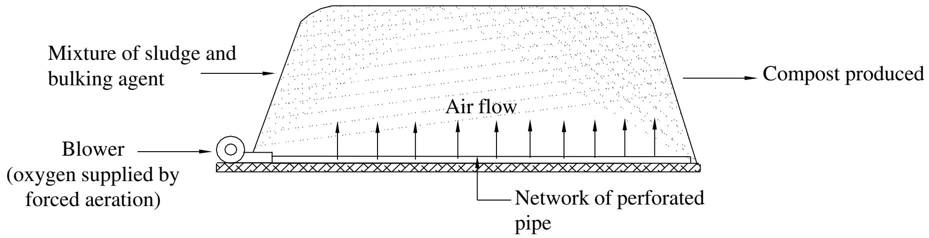


Figure 17. The Schematic Diagram of Aerated Static Pile Composting Method.

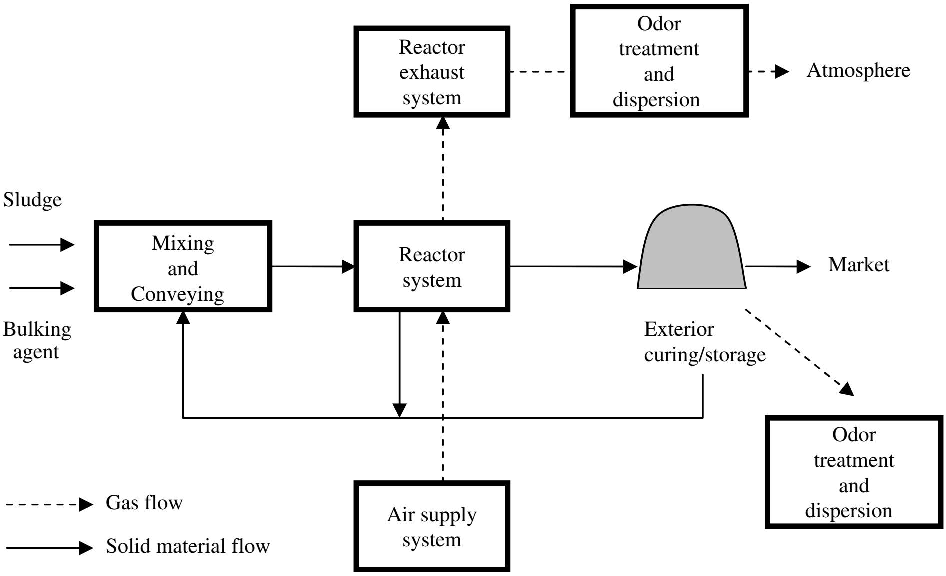


Figure 18. Flow Diagram of In-vessel Composting.[70]

agitator such as augers, conveyors, rams, or other devices.[69] The finished product is usually stored in a pile for additional curing prior to distribution after an active composting stage. Because of constant mixing, in-vessel composting has a great potential to produce a very uniform and high-quality compost. Basically, these mechanical systems are designed to minimize odor and shorten the composting period by controlling air flow, temperature, and oxygen concentration. Figure 18 describes the flow diagram of in-vessel composting.[70]

In-vessel composting systems can be divided into two major categories including plug flow (vertical and horizontal plug flows) and dynamic (agitated bin) reactors.[70] The primary differences between these reactors are the configurations of the aeration system and the discharge mechanisms. In a vertical plug flow reactor, the mixture of sludge and bulking agent is placed into the top of the reactor and moved to the bottom. Then, the compost is discharged by a rotating screw at the end of the reactor. The composting mixture is aerated but not agitated or mixed. Air is introduced either from the bottom or through a metal pipe supplying a jet of oxygen, which hanging from the top of the reactor.

Horizontal plug flow reactors are similar to vertical reactors in which the contents are not mixed within the reactor. In these reactors, the mixture (sludge and bulking agent) is loaded at one end of the reactor. The mixture is pushed through the reactor by a steel ram. Air is introduced through slots in the floor of the reactor. Final product (compost) is discharged from the end of the reactor, which is situated opposite the ram.

Table 35. The Advantages and Disadvantages of In-vessel Composting Method.[66,69,71]

Advantages
• The process is slightly affected by weather.
• Less space required.
• The quality of compost produced is more consistent.
• Not labor intensive.
• Good odor control because of self-containment.
• Good process control.
• Less workforce is required.
Disadvantages
• High capital cost.
• Highly dependent on mechanical equipment.
• Require careful management.
• Sensitive to changes in characteristics of biosolids and bulking agent.

In agitated bin reactors, the composting mixture (sludge and bulking material) is introduced from top. Basically, the difference between the plug flows reactor and the agitated bin reactors is that the compost does not move as an unmixed mass through the reactor. In this system, the composting mixture is mixed or periodically agitated by using mechanical device during the processing. Meanwhile, the air is supplied through the floor of the reactors. Physically, the reactors are open-topped bins.

Compared to other types of composting method, the in-vessel composting method offers the various advantages that are listed in Table 35. However, in-vessel composting method also has some disadvantages, which are also listed in Table 35.

3.2.4.3. Factors affecting composting process

Composting is simply the enhancement of the natural biological degradation of organic matter. Most composting process involving sludge or other biodegradable organic matters is done under aerobic conditions by microorganisms that require oxygen to survive. Microorganisms play a significant role to convert the organic matter into a humus-like material that is known as compost. Generally, in addition to compost, the primary by — products of aerobic composting are carbon dioxide, water, ammonia, heat, new cells, and resistant organic matter. The Eq. (11) illustrates the conversion of organic matter in the presence of oxygen:

$$\begin{aligned}\text{Organic matter} + O_2 + \text{nutrients} \longrightarrow CO_2 + H_2O + NH_3 + SO_4^{2-} + \text{heat}\\ + \text{new cells} + \text{resistant organic matter}\end{aligned} \tag{11}$$

Basically, before conducting any mixing, receiving, or material handling processes, it is important to consider various factors or elements that affect the composting process. Composting is influenced by five critical factors, which are:

(i) temperature,
(ii) moisture content,
(iii) nutrient (C:N ratio),
(iv) aeration or oxygen concentration, and
(v) pH.

Temperature plays a significant role in the composting process and greatly influences the microbial activity.[72] Basically, the desirable optimum temperature for composting is in the range of 50–70°C.[73,74] Higher temperatures are required to kill pathogens and weed seeds within the composting materials. However, excessive temperatures will slow or retard the composting process by causing many of the composting microorganisms to die off.

Based on the temperature changes that is caused by the microbial activity within the composting pile, the composting process generally divided into four different stages. They are mesophilic (first), thermophilic, mesophilic (second), and maturation/curing stages. Initial stage of decomposition process is carried out by mesophilic bacteria; this stage is known as mesophilic stage. The mesophilic bacteria exist in the temperature range of 25–45°C. A very large quantity of substrate (food) exists in the composting pile during this stage ensures that the microorganisms are very active, breaking down the soluble and easily degraded compounds, which lead to the generation of large quantities of metabolic heat energy. As the microorganisms give off heat, the temperature increases rapidly. In the second stage, the mesophilic microorganisms are then replaced by thermophilic microorganisms that thrive when the temperature rises above 45°C. High temperatures in the second stage will promote the decomposition of proteins, carbohydrates, and other organic compounds that provide important nutrients for the microorganisms. The temperature should rise rapidly to a peak between 55°C and 66°C within 1–4 days, and it should remain high for 1–3 weeks or more before decreasing.[75] In order to adequately inactivate pathogens and weed seeds, the temperatures throughout the compost pile must remain above 55°C for several days. However, when the temperature exceeds 65°C, most of the thermophilic microorganisms involved in the composting process begin to die off. At this condition, with the exhaustion of the food sources, microbial activity decreases and the temperature falls, rapid decomposition will stop, and the pile returns to the second mesophilic stage. When the temperature drops to 45–50°C, active composting is over. Mesophilic microorganisms again take over and complete the composting process in the maturation or curing stage. When monitoring the composting process, turning and aeration can be used to regulate temperature. Therefore, in the composting process, temperature management is important.

The second important factor that influences the composting process is moisture content. Moisture content of the composting material is an important environmental variable, as it provides a medium for the transport of dissolved nutrients required for the metabolic and physiological activities of microorganisms.[76] Composting works best if the moisture content of the composting material such as sludge is between 50% and 60%[72,76,77]; however, it would depend on the type of material. Too low moisture content (<10%) would cause dehydration during composting and this will slow the decomposition process. To overcome this problem, it is necessary to add water or material with a high moisture content. Besides that, high moisture content (>70%) in the compost pile will prevent oxygen diffusion to the microorganisms and this will produce anaerobic conditions. This also will lead to odor production and excess draining leachate. The possible way to overcome this problem is adding bulking material such as wood chips, straw, or leaves to lower the moisture content; indirectly, it will increase the pore space to allow oxygen to penetrate the compost pile better. Another way to overcome high moisture content in the compost pile is turning the pile frequently in order to increase the pore space and redistributes moisture from the inner part to outer space of composting pile. Generally, too low or too high moisture content in the composting pile will prevent and retard the ongoing composting process.

Microorganisms require carbon (C), nitrogen (N), phosphorus (P), and potassium (K) as primary nutrients. Of the primary nutrients, carbon and nitrogen play the most important role in the composting process. C:N ratio is referred to the amounts of carbon and nitrogen that are relative to one another. Basically, the C:N ratio is considered as one of the important parameters in composting because it provides a useful indication of the probable rate of decomposition of organic matters.[57] Carbon provides the fundamental backbone of organic molecules that supply energy for microbial activity while nitrogen is essential for the growth and development of microbial cell tissue. The most desirable optimum C:N ratio for composting varies between 20 and 25 parts of carbon to 1 part of nitrogen.[74,75] Few researchers also have mentioned another suitable optimum C:N ratios such as 25–30.[78] and 20–40.[79] However, both values are accepted because composting process is also influenced by other factors such as temperature, moisture content, and pH. The optimum C:N ratio can be obtained by combining or mixing various materials or organic waste. For example, digested sewage sludge with the C:N ratio of 16:1 can be blended with other carbon-rich waste (wood chips or straw) to increase the C:N ratio of the mixture to optimum levels.

Generally, the composting material with low C:N ratio (<20:1), which is used at the beginning of the composting process, can lead to the loss of nitrogen in the form of ammonia besides creating odor problem.[75,80] To overcome this problem, carbon-rich material such as wood chips and straw should be added. Meanwhile, where the

C:N ratio is higher (>35:1), the growth of microorganisms is limited, resulting in a longer composting time. Sometimes, higher C:N ratio may retard the composting process.[80] The addition of nitrogen-rich waste such as sludge and manure can help to obtain the optimum C:N ratio. It has been stated that when the C:N ratio is less than 20:1, the compost is considered mature and can be used without any restrictions on plant or crop.[81] A final compost with a C:N ratio of higher than 20:1 should be because it could have a negative impact on plant growth and seed germination.

Aerobic composting requires large amounts of oxygen, especially at the initial stage. Aeration is the source of oxygen and thus it is absolutely necessary for aerobic composting. Limiting the oxygen supply to the organic materials slows down the composting process, creating anaerobic conditions and potential odors. The functions of aeration in the composting process include meeting stoichiometric oxygen demands for organic decomposition, removal of excess moisture in wet substrates, and temperature control by transferring generated heat out of the compost pile.[64] Basically, in the composting process, low oxygen levels could lead odors when the compost pile is broken down. Meanwhile, excessive oxygen levels tend to keep temperatures low and they slow the rate of composting. At certain circumstances, inadequate airflow may cause excessive temperatures in the compost pile. Heat removal is important in warm climates as the risk of overheating and fire is higher. Therefore, good aeration is necessary for efficient composting. Aeration can be achieved through natural or forced convection or through mixing of the material such as turning or agitation. Normally, the lowest oxygen concentrations are found at the center and at the base of the compost pile.[75] There are several factors that should be considered to achieve good aeration in the composting pile such as particle size, moisture content, type of material, pile size, ventilation, and frequency of turning.

Composting can occur over a wide pH range due to a variety of microorganisms involved. However, the optimum pH range preferred is normally between 6.5 and 8.5.[82] During the composting process, pH initially drops due to the formation of organic acids and as the composting proceeds, the pH becomes neutral again. This is because these acids are converted to methane and CO_2. However, high pH (above 8) in compost mixtures should be avoided because this condition will result in ammonia volatilization and odor problems.[78] As the compost matures, the pH of final compost is usually slightly alkaline with the range of 7.5–8.5.

3.2.4.4. Advantages and disadvantages of composting process

In general, various advantages have been claimed for composting process. The potential advantages of this stabilization method are listed in Table 36. However, this process also has some disadvantages and these are also listed in Table 36.

Table 36. The Advantages and Disadvantages of Composting Process.[66,72]

Advantages

- The process can be started with very little capital and operating costs.
- Ease of storage, handling, and use of composted product.
- Addition of compost produced from the process can help to increase phosphorus, potassium, nitrogen, and organic carbon contents of the soil.
- Emphasis on beneficial reuse at federal, state, and local levels.
- Compost produced can also be used in various land applications such as for wetland and mine land restoration, bioremediation of hazardous sites, and pollution prevention.
- Compost produced has a potential to alter or degrade many types of contaminants such as wood preservatives, solvents, heavy metals, pesticides, petroleum products, and explosives.

Disadvantages

- Poor feed stock, which yields poor quality finished compost; for example, heavy metal contamination.
- Nuisance potential such formation of odor (if anaerobic conditions develop) at the composting site and the presence of rats.
- Survival and presence of primary pathogens in the products.
- Lack of consistency in product quality with reference to metals, stability, and maturity.
- Composting facilities require more space. It often related to storage and market demand.
- A product must be marketed.

3.2.5. *Heat Drying*

3.2.5.1. General description

Generally, heat drying is one of the methods used for sludge stabilization where it plays an important role to remove the moisture from wastewater solids; as a result, it can be incinerated efficiently or processed into fertilizer. This method is used to destroy pathogens and eliminate most of the water content and at the same time it reduces the volume of sludge. In sludge management, heat drying has been employed as one of the options to reduce sludge volume and at the same time producing a final product that can be used beneficially. Heat drying also has a great potential to produce Class A biosolids.[83] Class A biosolids contain minute levels of pathogens. Heat-dried product produced after the heat-drying process that have met the pathogen reduction requirements can be land applied without any pathogen-related restrictions at the site. Besides that, it can be sold or given away for residential use. Several reasons have been listed for employing heat drying including.[1]

- to market the final products in bags,
- to increase the fuel value of sludge, and
- to reduce sludge transportation costs.

Historically, heat-drying method has been employed since 1920s in the United States to dry WAS and sludge produced from the Imhoff tank in order to produce a soil

Table 37. Examples of Wastewater Solids Dryers in the United States.[83]

Location	Type of Dryer	Type of Biosolids Dried
Milwaukee, WI	Direct rotary dryer	Blend of raw secondary with digested primary
Baltimore, MD (Patapsco)	Direct rotary dryer	Blend of raw secondary with digested secondary
North Andover, MA	Direct rotary dryer	Anaerobically digested
Newport, TN	Direct rotary dryer	Anaerobically digested
Sacramento, CA	Direct rotary dryer	Anaerobically digested
Ocean County, NJ	Direct rotary dryer	Anaerobically digested
Waco, TX	Direct rotary dryer	Anaerobically digested
New York City, NY	Direct rotary dryer	Anaerobically digested
Amsterdam, NY	Direct rotary dryer	Anaerobically digested

conditioner.[5] In the mid-1980s, this technology has developed, and has been used by many wastewater solids generators. Table 37 shows the example of wastewater solids dryers used in the United State.

3.2.5.2. Heat-drying stages

Sludge is heat dried at temperatures too low to destroy organic matter. Water vapor is carried away by air. In designing heat-drying process, the process engineer has to establish the actual conditions of drying such as temperature, humidity, detention time, velocity, and direction of the gas flow. Basically, there are three well-defined stages involved in heat drying, which are initial drying, steady-state drying, and final drying (Table 38).

3.2.5.3. Types of dryers

Dryers can be classified as direct, indirect, or other.[83] Direct and indirect dryers typically have been most successful for drying wastewater solids. Basically, in direct dryers, the wastewater solids come into contact with hot influent gases. As a result, the moisture evaporated from the surface of sludge. There are several types of direct dryers used nowadays including rotary dryers, flash dryers, spray dryers, and toroidal dryers. The end product (pellets) produced from the direct dryers are usually uniform in size, texture, and durability. Because of these characteristics, they seldom require additional processing to make them marketable. Generally, the plant must mix processed solids into the feed solid to raise solids content of the feed mixture and avoid a condition referred to as the “sticky” or “plastic” phase.

In indirect dryer system, the solids are separated from the heating medium by a retaining wall (normally metal wall). At these conditions, the solids never come into

Table 38. Stages of Heat Drying of Sludges.[5]

Stages	Information
Initial drying (Stage 1)	During this stage, the sludge temperature and the drying rates are increased to the steady-state conditions of the second stage. Stage 1 is usually short; little drying occurs during this time.
Steady state drying (Stage 2)	The time that the sludge is in this stage is generally the longest of all the stages. The surfaces of the sludge particles are completely saturated with water. Surface water is replaced with water from the interior of the solid as fast as it is evaporated. Drying proceeds as if the water was evaporated from a pool of liquid. The solid itself does not significantly influence in the drying rate. During this stage, the temperature at the sludge/gas interface is ordinarily kept at the wet bulb temperature of the gas.
Final drying (Stage 3)	The final stage occurs when sufficient water has evaporated that the solid surface is only partially saturated. Surface water is evaporated more rapidly than it can be replaced by water from the interior of the solid. As a consequence, overall drying rates are markedly lower in Stage 3 than in Stage 2. During this period, the temperature of the solid/gas interface increases because latent heat cannot be transferred from the sludge to the gas phase as rapidly as sensible heat is received from the heating medium.

Table 39. The Advantages and Disadvantages of Indirect Dryers.[83]

Advantages

- Have higher thermal efficiency and are more suitable when pellets are to be used in energy production or combusted.
- Produce less dust during the drying process.
- Have lower risk of explosion than direct dryers.

Disadvantages

- Heat-dried product of indirect dryers tends to be dustier than a dried product from a direct dryer.
- Often produce oversized pellets.
- Might require additional processing such as granulation or compaction to increase uniformity, consistency, and durability of the heat-dried end product.

direct contact with the heating medium. Moisture evaporates when the wastewater solids contact the metal surface heated by the hot medium. Normally, the heat transfer surface consists of a series of hollow metal discs or paddles mounted on a rotating shaft, through which the heating medium flows. The functions of rotating shaft are to agitate solids, improve heat transfer, and facilitate the movement of solids through the dryer. In some indirect drying systems, mixing of previously dried material with feed solids is required. Examples of indirect dryers include steam dryers, hollow flight dryers, and tray dryers. Indirect dryers have several advantages and disadvantages in sludge heat drying system compared to direct dryers, which are listed in Table 39.

Table 40. The Advantages and Disadvantages of Heat Drying Process.[83]

Advantages
• Requires small space compared to other stabilization process such as composting, alkaline stabilization, and air drying. • This method can be designed to accept a variety of feed material characteristics. • Reduces the volume of material that needs to be transported. • Reduces traffic into and out of a facility. • Generates a readily marketable product.
Disadvantages
• Requires large capital cost. • Requires large amount of energy. • Generates dust that can affect workers and neighbors in the local community. • Creates an explosive hazard from dust generated in the drying process. • Requires skilled workers for handling operation and maintenance. • Produce nuisance odors.

3.2.5.4. Advantages and disadvantages of heat drying

Heat drying offers several advantages in sludge stabilization. Despite the well-known advantages, there are some disadvantages compared to other stabilization processes and these have been tabulated in Table 40.

3.3. *Conditioning*

3.3.1. *General Description*

Sludge particles contain enormous amount of bound water (the state of water in many porous substances that retain moisture). To release these bound water and to allow agglomeration of solids, a preliminary conditioning stage is required prior to the dewatering process. The following lists describe the functions of sludge conditioning before dewatering process[35]:

- flocculation of particularly fine SS,
- washing out the alkalinity of anaerobically digested sludge,
- promotion of rapid formation of a stable drainable cake,
- promotion of cake release from filtration support media,
- enhancement of cake fuel value, and
- prevention of scale formation and corrosion inhibition.

Conditioning can be defined as a process whereby wastewater solids are treated with chemicals/physical particles or various other means to enhance water removal and to improve solid capture.[1] In addition, some conditioning processes help to disinfect sludge, control odors, alter the nature of solids, provide limited solids destruction, and improve solids recovery.[5] Several methods of conditioning sludge are available nowadays such as chemical conditioning (involving the use of organic chemicals and inorganic chemicals) and heat treatment. Besides these two methods, there are several other methods that can be employed for sludge conditioning, which include freezing, irradiation, and solvent extraction. However, only chemical and thermal conditionings will be discussed in depth in this section.

3.3.2. *Chemical Conditioning*

Basically, chemical conditioning is associated with mechanical sludge dewatering systems including vacuum filter, filter press, belt filter, and centrifugation. Chemical conditioning is achieved by the addition of organic or inorganic chemicals to the wastewater sludge.[84] The use of chemical for this purpose is economical because it can increase yields and obtain greater flexibility. Chemical conditioning has a great potential in reducing 90–99% of sludge moisture content to a lower level of 65–85%.[4] However, the percentage of reduction is depended on the nature of wastewater solids to be treated. The addition of the chemical to the sludge lowers or raises its pH value to a point where small particles coagulate into larger ones. There is no one pH value best for all sludges. Different sludges have different optimum pH values, and it must be determined for each sludge prior to further treatment. Generally, chemicals are most easily used in liquid form. If the chemicals are in dry powder form, dissolving tank is needed for preparing the solution. The dissolving tank must be lined with rubber or corrosion-resistant material. The tank is usually made of polyvinyl chloride, polyethylene, and rubber.

3.3.2.1. Inorganic chemical conditioning

Inorganic chemical conditioning is associated principally with mechanical sludge dewatering especially vacuum filtration. The chemicals that are commonly used in the conditioning of sludge are ferric chloride and lime.[5] However, ferrous sulfate also can be used to replace ferric chloride and lime. In the conditioning process, ferric chloride and lime are normally used in combination. However, it is not unusual for them to be applied individually. For instance, lime alone is a quite popular conditioner for primary sludge whereas ferric chloride alone has been used for conditioning activated sludge.

At the beginning of the conditioning process, ferric chloride is normally added first. It hydrolyzes water and forming positively charged soluble iron complexes. The iron complexes then neutralize the negatively charge sludge solids, which causing them to combine together. Ferric chloride also reacts with bicarbonate alkalinity in the sludge to form hydroxides that act as flocculants. Equation (12) illustrates the

Table 41. Typical Conditioning Dosages of Ferric Chloride and Lime for Municipal Wastewater Sludges.[85]

	Vacuum Filter		Recessed-plate Pressure Filters	
Sludge Type	Ferric Chloride	Lime	Ferric Chloride	Lime
Raw primary	40–80	160–200	80–120	220–280
Raw WAS (WAS)-air	120–200	0–320	140–200	400–500
Raw (primary + trickling filter)	40–80	180–240		
Raw (primary + WAS)	50–120	180–320		
Raw (primary + WAS + septic)	50–80	240–300		
Raw (primary + WAS + lime)	30–50	None		
Elutriated anaerobically digested				
Primary	50–80	0–100		
Primary + WAS (air)	60–120	0–150		
Thermal conditioned sludges	None	None	None	None
Anaerobically digested sludges				
Primary	60–100	200–260		
Primary + trickling filter	80–120	250–350		
Primary + WAS (air)	60–120	300–420	80–200	220–600

[a] All values shown are for pounds of either ferric chloride or lime per ton of dry solids pumped to the dewatering unit.
1 lb/ton = 0.5 kg/ton

reaction of ferric chloride with bicarbonate alkalinity.

$$2FeCl_3 + 3Ca(HCO_3)_2 \longrightarrow 2Fe(OH)_3 + 3CaCl_2 + 6CO_2 \tag{12}$$

Hydrated lime is usually combined with ferric iron salts. Lime has a great potential to provide pH control, odor reduction, and disinfection. $CaCO_3$, the product formed in the reaction of lime and bicarbonate, provides a granular structure that increases sludge porosity and reduces sludge compressibility. Iron salts are usually added at a dosage rate of 20–63 kg/ton of dry solids in the sludge feed, whether or not lime is used. Lime dosage usually varies from 75 to 277 kg/ton of dry solids fed. Table 41 shows typical ferric chloride and lime dosages for various types of sludges used in two different dewatering units.

Beside ferric chloride and lime, other inorganic chemical conditioners that can be employed in the chemical conditioning include pulverized coal, cement kiln dust, fly ash, power plant ash, and sludge incinerator ash.[86]

3.3.2.2. Organic chemical conditioning

Historically, organic chemical conditioners were first used in the 1960s.[63] Unlike inorganic chemical conditioners, which are only a few in numbers to consider, the organic chemical conditioners consist of various products that differ greatly in chemical composition, functional effectiveness, and cost effectiveness[5].

Organic polymers or polyelectrolyte are widely used in wastewater treatment and sludge conditioning. They are long-chain, water-soluble, and specialty chemicals. In sludge conditioning, the most commonly used polymers are the cationic polymers. They can be synthesized from individual monomers or can be made by the chemical addition of functional monomers, or groups to naturally occurring polymers. A monomer is the subunit from which polymers are made through various types of polymerization reactions. The backbone unit or monomer most widely used in synthetic organic polyelectrolyte, which is known as acrylamide. By the year 1979, synthesized polymers are widely used. Polyacrylamide is created when the monomers combine to form long and thread-like molecule with molecular weight in the millions. The charge type of the polyelectrolyte refers to the positive or negative electrical charge sites along the polymer chain. Anionic type polyacrylamide flocculants carry a negative electrical charge in aqueous solutions. They are made by either hydrolyzing the amide group (NH_2) or combining the acrylamide polymer with an anionic monomer. Meanwhile, cationic polyacrylamides carry a positive electrical charge in aqueous solutions. These polyelectrolytes can be prepared by chemical modification of nonionic polyacrylamide or by combining the cationic monomer with acrylamide. In sludge conditioning, cationic polyelectrolytes are most widely used polymers since most sludge solids carry a negative charge. Polyamines, polyethylimines, polyimidamines, polybutadienes, polyamidamines, polyquaternaries, and substituted polyacrylamides are the examples of cationic polymers that are widely used in sludge conditioning.

Generally, polymers used for sludge conditioning are manufactured in five different forms, which are dry, emulsions, liquid, mannich, and gel forms.[63] Cationic polyelectrolytes are available in dry powders or liquids form. Usually, liquids come as water solutions or emulsions. The shelf life of the dry powders is usually several years while most of the liquids have shelf lives of two to six months. Organic polyelectrolytes that are dissolved in water tend to form solutions with varying viscosity. The resulting viscosity is normally depends on their molecular weight and degree of ionic charge.

There are several reasons that influence the selection of organic polymers over inorganic chemical conditioners as the conditioning agent. These include[63]:

- Less dosage should be added for conditioning process.
- Produce little additional sludge mass and volume.
- Do not lower the fuel value of the sludge cake if incineration to be used.
- Allow for cleaner material handling operations.
- Reduce operation and maintenance problems.
- More effective in terms of achieving a high solids recovery and producing a cake with a high solids content.

3.3.3. *Thermal Conditioning*

3.3.3.1. General information

Thermal conditioning subjects the sludge particles to heat at elevated pressures in a confined reactor to enhance the dewatering without the addition of chemicals. In thermal conditioning, the application of heat breaks up cell walls of microorganisms contained in biological sludges and releasing bound water from sludge particles. The heat treatment results in coagulation of solids, decomposition of large quantities of cell mass, and reduction of water affinity of sludge solids. By definition, thermal conditioning or also known as heat treatment involves heating of wastewater sludge, with or without the addition of air or oxygen to a temperature of about 350°F to 400°F (177°C to 240°C) in a reactor under pressures of 250–400 psig (1723–2758 kN/m^2) for about 15–40 min.[5] This process causes the release of water and organic material from sludge in the form of a dark brown fluid.[35] Figures 19 and 20 depict a flow diagram for a typical thermal sludge conditioning system for plants without and with

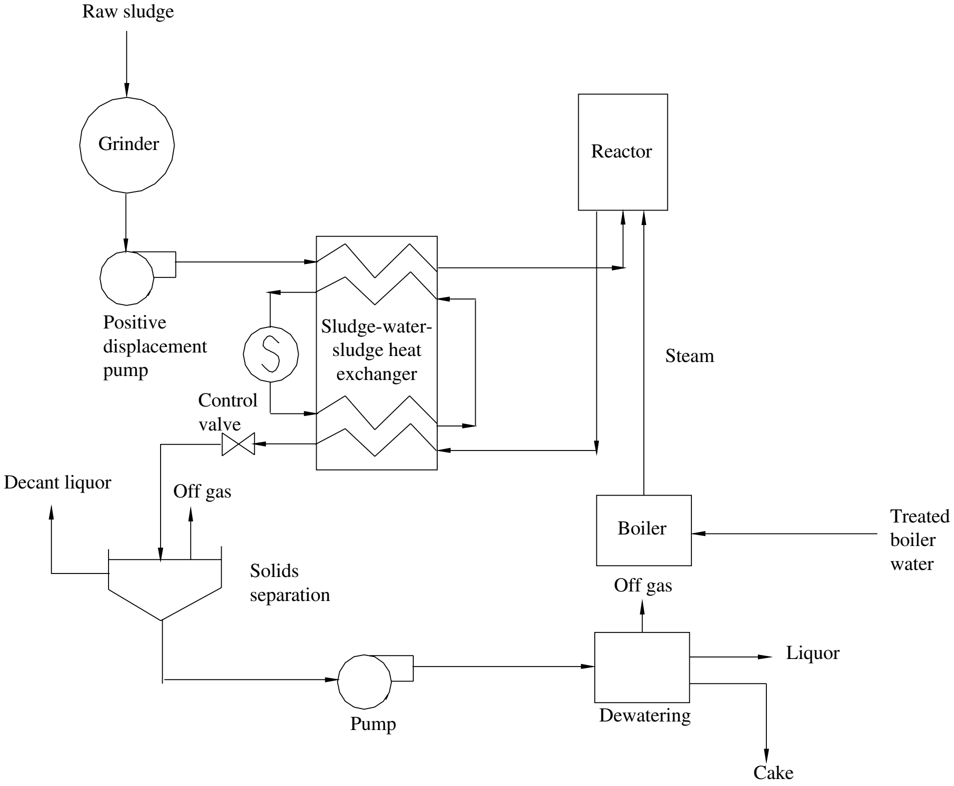


Figure 19. Flow Diagram for a Typical Thermal Sludge Conditioning System for Plants Without the Addition of Air.[5]

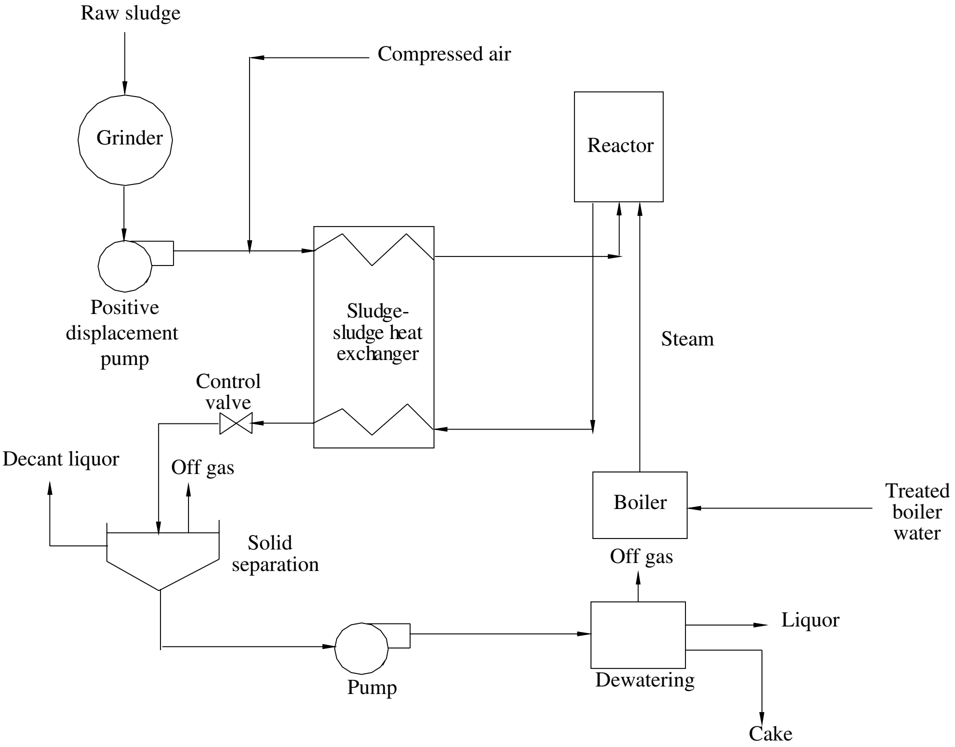


Figure 20. Flow Diagram for a Typical Thermal Sludge Conditioning System for Plants with the Addition of Air.[5]

the addition of air. Typically, the major components in the system are heat exchanger and reactor.

3.3.3.2. Historical background

Thermal conditioning as a process for pretreating sludge has a long history. In the mid-1930s, thermal conditioning was first studied by William K. Porteous in England. Meanwhile, thermal conditioning in the United States was first studied in the mid-1960s. In 1969, the first facility that was installed for plant without the addition of air is located at Colorado Springs, Colorado whereas the first plant with air addition was installed at Levittown, Pennsylvania in 1967. Since then, almost over 100 thermal sludge-conditioning installations have been built in the United States.[5]

3.3.3.3. Advantages and disadvantages of thermal conditioning

Thermal conditioning has several advantages and disadvantages. Both advantages and disadvantages of this system are listed in Table 42.

Table 42. The Advantages and Disadvantages of Thermal Conditioning.[5]

Advantages

- The process will produce a more readily dewaterable sludge compared with chemical conditioning. The solids content of dewatered sludge can range from 30 to 50%.
- The process is relatively insensitive to changes in sludge composition.
- The process does not require length or elaborate start up procedures.
- The end product normally does not require chemical conditioning.
- The process sterilizes the sludge, providing it free of pathogenic organisms.
- The processed sludge have a heating value of 12,000–13,000 Btu/pound (28–30 kJ/g) of volatile solids.
- The process is appropriate for large number of sludges that cannot be stabilized biologically because of the presence of toxics materials.

Disadvantages

- The process has high capital and maintenance costs.
- The process requires close supervision, skilled operators, and a strong preventive maintenance program.
- The process produces odorous gas stream that must be collected and treated before release to the environment.
- The process produces liquid sidestreams with high concentration of BOD, ammonia nitrogen, and color as a result of solubilization of some of the organic solids in the sludge being treated.
- Formation of scale and acid-induced corrosion in heat exchangers, pipes, and reactors.

3.3.4. *Factors Affecting Conditioning Process*

Conditioning always has a significant effect on the efficiency of the thickening or dewatering process. In general, all characteristics relate to the difficulty of forcing sludge solids closer together, or to the difficulty of water movement through the voids between the sludge solids. Sludge conditioning is purposely done to counteract adverse characteristics, which decrease the rate or degree of water removal. The several factors that influence conditioning process are[23]:

- particle surface charge and dehydration,
- particle size,
- compressibility,
- sludge temperature,
- ratio of volatile solids to fixed solids,
- sludge pH, and
- septicity.

3.3.4.1. Particle surface charge and dehydration

Generally, sludge exclude from the text particles have a negative surface charge and, therefore, they repel each other. As a result, the repulsive force increases when they are forced closer. Additionally, sludge particles retain water molecules to their surface either by adsorption or by capillary action. To overcome the effects of surface charge and surface hydration, chemical conditioners such as lime, ferric chloride, or organic polymers are added into the tank. These chemicals act by reducing or eliminating repulsive force, thus allowing the particles to flocculate. This action will facilitate water removal in the subsequent dewatering process.

3.3.4.2. Particle size

Particle size is generally considered as the most significant factor affecting dewaterability. When the average size of particle decreases, the surface area for a given sludge mass increases. The increasing of surface area gives the following effects:

- Greater electrical repulsion between sludge particles due to a larger area of negatively charged surface.
- Increased frictional resistance to the movement of water.
- Increased attraction of water to the particle surface due to more adsorption sites.

Sludge particle size is influenced by sludge source and prior treatment. Primary sludge generally has a larger average particle size than secondary sludge. This is because fine and colloidal solids tend to pass through the primary clarifier. The fine particles generally are removed in the secondary clarifier together with the less dense and flocculated cellular material that is produced during biological treatment. Particle size is also decreased when the primary or secondary sludge is stabilized under aerobic or anaerobic treatment. This is the main reason that digested sludge is more difficult to dewater than raw sludge. Besides that, sludge particle size can also be decreased through mixing, storage, and sludge transport.

3.3.4.3. Compressibility

Sludge compressibility results in particle deformation and a reduction in the void area between particles. The reduction in void volume prevents the movement of water through the compressed portion of the sludge, thus reduces the rate of dewaterability. Proper conditioning helps to improve dewaterability by producing a flocculant matrix of solids in relatively clear water before the initiation of filtration. Therefore, the net result of conditioning is a more rapid removal of water.

3.3.4.4. Sludge temperature

The temperature of sludge is indirectly related to the viscosity of the water present in the sludge mass. When the temperature of sludge increases, the viscosity of water

Table 43. Viscosity of Water as a Function of Temperature.[23]

Temperature (°C)	Viscosity (Centipoise)
10	1.308
15	1.140
20	1.005
25	0.894
30	0.800
35	0.723

Note: 1 centiPoise = 0.001 Pascal seconds.

in the sludge mass decreases. Viscosity is particularly important in the centrifugation. Table 43 illustrates the relationship between water temperature and viscosity of water.

3.3.4.5. Ratio of volatile solids to fixed solids

Sludge is easier to dewater, as the percentage of fixed solids increases relative to volatile solids provided that all other factors are equal.

3.3.4.6. Sludge pH

pH affects the surface charge on sludge particles as well as influences the type of polymer to be used for conditioning. Generally, anionic polymers are most useful when the sludge is lime conditioned and has a high pH, while cationic polymers are most suitable at pH slightly above or below neutral values.

3.3.4.7. Septicity

Septic sludge is more difficult to dewater than fresh sludge. As a result, it requires more chemical conditioners (in higher dosages) to condition and dewater the sludge.

3.4. *Dewatering*

3.4.1. *General Description*

Most of all wastewater treatment facilities require a means of handling and disposing of sludges generated by the treatment processes. The sludge handling process involves the dewatering of the liquid sludge to reduce the sludge volume and produce a relatively dry sludge cake for additional treatment or less costly treatment and disposal. Dewatering can be defined as the removal process of water from sludge solids to achieve a volume reduction greater than that achieved by thickening and indirectly increasing the solids concentration. Dewatering sludge from 5% to 20% solids concentration reduces volume by 3/4 and produces a nonfluid material.[5] Before dewatering process being carried out, sludge is usually conditioned and thickened. In conditioning, chemicals such as ferric chloride, lime, or polymers are added to facilitate the separation of solids by aggregating small particles into larger flocs; while in thickening, a part of the water bound to sludge particles is removed to concentrate the solids material.

Dewatering process is used to reduce moisture content of sludge for several reasons, which include[2,4]:

- Dewatering helps to reduce transportation costs and makes handling to the ultimate disposal sites easier because sludge is already converted from liquid form to a damp cake.
- Dewatered sludge is easier to handle than liquid sludge. This is because they may be shoveled, move about with tractors fitted with buckets and blades, and transported by conveyor belts.
- Dewatering is required for sludge destined for incineration to prevent damage to boilers and decrease the energy required for combustion.
- Dewatering is often a necessary process before treatment or use of the processes such as composting, drying, or land application.
- Dewatering is necessary to remove excess moisture in order to prevent odor production.
- Dewatering is required before landfilling in order to prevent leachate production. Disposal of liquid in landfill is prohibited.

3.4.2. *Classification of Dewatering Methods*

Typical dewatering methods are classified into two types, which are air drying (drying beds) and mechanical dewatering systems.[5] Briefly, air drying involves placing the sludge on a sand bed and allowing them to dry through evaporation and drainage. This process can produce a solid content in primary biosolids of as high as 45–90%. This process is relatively simple in terms of operation; however, it requires large land areas and relatively long periods of time. Normally, this system tends to be used by small- and medium-sized treatment plants that generate small amount of sludge. Meanwhile, mechanical dewatering systems such as vacuum filters, plate and frame filter presses, belt filter presses, and centrifuges are widely utilized especially in large treatment plants.

This section will discuss further on drying beds (sand, paved, wedge wire, and vacuum-assisted drying bed) and mechanical sludge dewatering systems (vacuum filters, belt filter presses, and recessed-plate filter presses). However, centrifuges will not be discussed in this section since it has been discussed in depth in Sec. 3.1.2.

3.4.3. *Air Drying System (Drying Beds)*

These systems are the most widely used method of municipal sludge dewatering in the United States.[5] Sludge drying beds are typically used to dewater digested sludge. Although the use of drying beds might be expected in smaller WWTPs and a warmer sunny region, they are also used in several large facilities in northern climates. Drying beds generally consist of a 1–3-ft-high (0.3–1.0-m) retaining wall enclosing a porous drainage media. This drainage media may be made up of various sandwiched layers of sand and gravel, combinations of sand and gravel with cement strips, slotted metal media, or permanent porous media. There are several appurtenant equipment including: sludge feed pipelines and flow meters, possible chemical application tanks, pipelines and metering pumps, filtrate drainage and recirculation lines, possible mechanical sludge removal equipment and a possible cover or enclosure. Table 44 depicts the advantages and disadvantages of drying bed method.

Basically, common operational procedures to all types of drying beds involve the following[5]:

- Stabilized liquid sludge (8–12 in. $\approx$ 20–30 cm) is pumped onto the drying bed surface.
- Chemical conditioners are added continuously. Conditioners should be injected into the sludge as it is pumped onto the bed.
- When the bed is filled to the desired level, the sludge is leaving to dry to the desired final solids concentration. The concentration can vary from 18% to 60% and depending on the type of sludge, processing rate needed, degree of dryness required for lifting, etc.
- The dewatered sludge is removed either mechanically or manually.
- Repeat the cycle.

Table 44. The Advantages and Disadvantages of Using Drying Beds.[5]

Advantages
• The method requires the lowest capital cost. • Requires little operator attention and low skilled labor. • Low energy consumption. • Less sensitive to sludge variability. • Require less chemical consumption. • Higher dry cake solids contents than mechanical methods.
Disadvantages
• Requires more land than mechanical method. • Requires a stabilized sludge. • Must consider the weather or climate effect when designing the system. • Removal usually labor intensive.

3.4.3.1. Conventional sand drying beds

Generally drying beds can be classified as conventional, paved, wedge wire, and vacuum-assisted drying bed. Conventional sand drying beds are known as the oldest and most commonly used type of drying bed. Normally, sand drying beds are constructed in rectangular shape with dimensions of 15–60 ft (4.5–18 m) wide and 50–1500 ft (15–47 m) long with vertical side walls. Sand is placed over graded gravel or stone. The thickness of the sand layer and gravel layer is usually in the range of 4–9 in. (10–23 cm) and 8–18 in. (20–46 cm), respectively. The diameter of sand used is usually in the range of 0.012–0.05 in. (0.3–1.2 mm) and has a uniformity coefficient less than 5.0. Meanwhile, the diameter of gravel is normally from 0.125 to 1.0 in. (0.3–2.5 cm). Underneath piping is made of vitrified clay; however, plastic pipe is also acceptable to be used. The pipes should be more than 4 in. (10 cm), 8–20 ft (2.4–6.0 m) spaced apart and placed at a minimum slope of 1%.

In this method, sludge is applied onto a layer of sand and gravel across the entire bed and allowed to drain and dry until the sludge is caked (dried) and cracked. The time taken for the sludge to be dried varies in a nonlinear manner with the thickness of the sludge applied. Generally, caking and cracking occur when the solids content reaches 35–40%.[23] Most sludge is removed when it reaches this content. The dried sludge can be removed either manually or mechanically (front-end loader). The removal process depends on the bed thickness. The use of mechanical equipment can cause problems because of its weight. A portion of sand might be lost when the sludge is removed. Periodic sand replenishment is necessary for the next drying process. Figure 21 depicts the typical sand drying bed construction.

Tables 45 and 46 list the advantages and disadvantages of sand drying beds and their common design shortcomings.

3.4.3.2. Paved drying beds

Paved drying beds have had limited use since 1954.[5] This method has been employed to overcome the problem that is related to the use of mechanical sludge removal equipment damaging the underneath pipes. Normally, paved beds are rectangular in shape with vertical side walls. The dimension of the bed is 20–50 ft (6–15 m) wide and 70–150 ft (21–46 m) long. Until recently, paved beds used an asphalt or concrete pavement on top of a porous gravel subbase.[87] The lining is normally placed on 8–12 in. (20–30 cm) sand or gravel base layer. The lining should have approximately 1.5–2.0% slope to the drainage area. A 4 in. (10 cm) pipe is located in the center beneath a sand drainage strip would convey drainage away. An unpaved area, which is 2–3 ft (0.6–1.0 m) wide is placed around the perimeter bed is functioned to collect and convey drainage water. The typical paved drying bed construction is illustrated in Fig. 22. The main advantage of this type of bed is the capability to use heavy

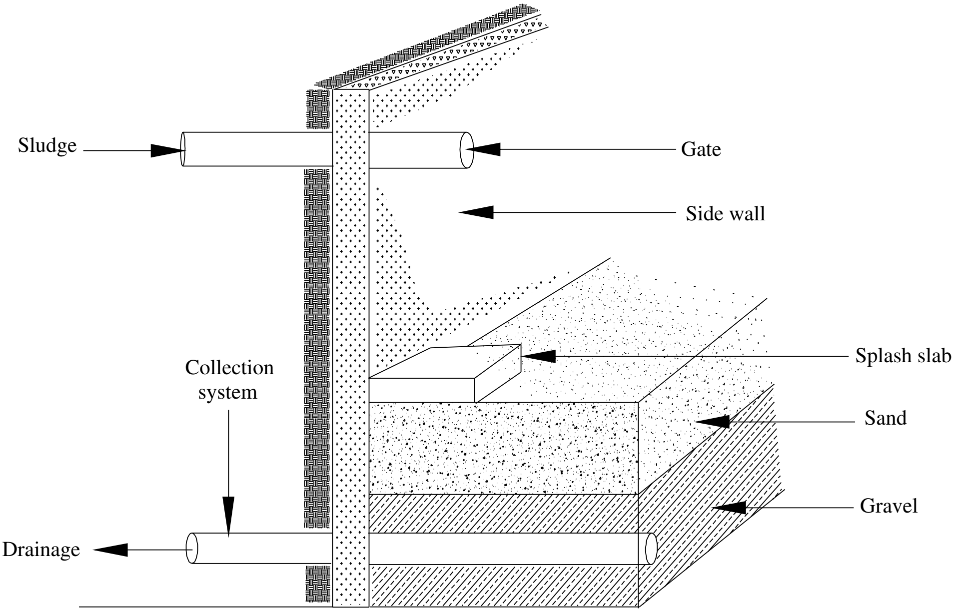


Figure 21. Typical Sand Drying Bed Construction.[5]

Table 45. The Advantages and Disadvantages of Using Sand Drying Beds.[23]

Advantages

- Requires low capital costs.
- Requires low operational/skill labor.
- Requires low maintenance material cost.
- Requires little or no chemical addition.
- Possible to produce high cake solids content.

Disadvantages

- Can be influenced by weather condition such as rainfall or cold weather.
- Requires large space.
- Requires many workers for sludge removal.
- Tend to produce odor if poorly stabilized sludge is used.
- May be aesthetically unpleasing and depends on location.

mechanical equipment for sludge removal without damaging the underneath pipes or loss of sand and reduced bed maintenance.[5,23,87] Besides that, this method also has disadvantages; for example, it requires high-capital cost and needs larger space than sand drying beds. Paved beds are applicable and can be used in any location. However, the concept is most advantageous in warm, arid, and semi-arid climates since the major pathway for water loss to the atmosphere is through evaporation.[87]

Table 46. Common Design Shortcomings of Sand Drying Bed Installations.[23]

Shortcomings	Resultant Problems	Solution
Inadequate bed area	Sludge need to be removed before it is dry enough. As a results, chemical conditioners may be required.	Construct additional beds or add chemical conditioners.
Inadequate access for removal of dried sludge	Dried sludge must be removed considerable distance to reach hauling truck.	Several options can be implemented such as: • construct roadway between beds, • cast concrete treadways in beds for vehicle access, and • use planks on bed to support vehicles.
Inadequate drainage system	Longer than necessary drying time.	Additional drainage pipes should be added.
Poor sludge distribution on the beds	Inadequate use of bed area.	Several options can be implemented such as: • construct smaller beds instead of large beds and • level sand in beds.
Improper sand gradation	Slow drainage.	Remove and replace sand.

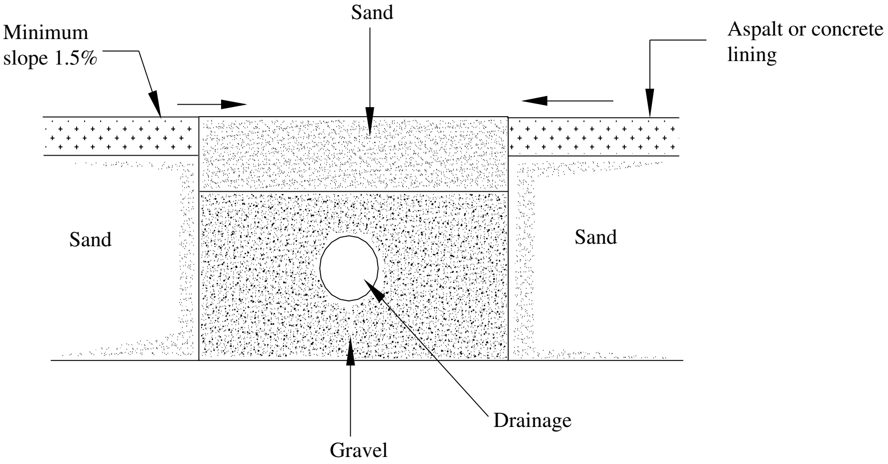


Figure 22. Typical Paved Drying Bed Construction.[5]

3.4.3.3. Wedge wire drying beds

Wedge wire drying beds is designed to introduce sludge slurry onto a horizontal relatively open drainage media in a way that would yield a clean filtrate and provides a reasonable drainage rate. This system was developed in England and has been successfully employed for over 20 years to dewater sludge produced from municipal and industrial wastewater treatment. Since 1970s, this system has been used widely in the United States. Normally, the material used for drying bed comprises of stainless steel wedge wire or high-density polyurethane.[23,88] Basically, the bed comprises of a shallow rectangular watertight basin fitted with a false floor of wedge water panels. These panels (wedge wire septum) have slotted openings of 0.01 in. (0.25 mm). The false floor is made watertight with caulking where the panels abut the walls. An outlet valve that is responsible to control the rate of drainage is located underneath the false floor. Figure 23 illustrates the cross section of a wedge wire drying bed.

The operating sequences for dewatering sludge by using wedge wire drying bed are as following[5,35]:

- The water or treatment plant effluent is introduced into the wedge water unit to reach a depth of 1 in. (2.5 cm). This water serves as a cushion that allows the added

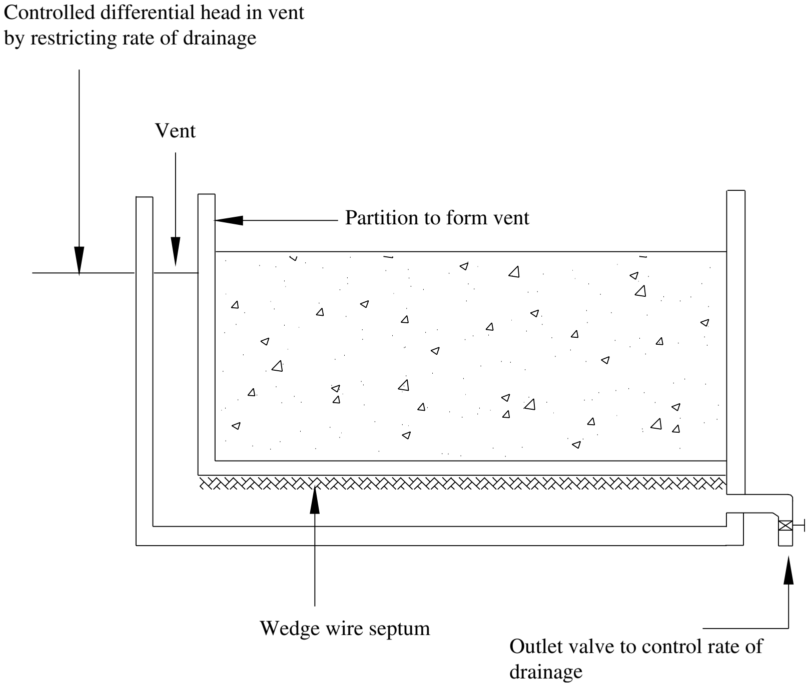


Figure 23. The Cross Section of a Wedge Wire Drying Bed.[5]

Table 47. The Advantages and Disadvantages of Using Wedge Wire Drying Beds.[5]

Advantages
• No clogging of the media. • Constant and rapid drainage. • Higher throughput rate than sand drying beds. • Ease of operation and maintenance. • Treatability of aerobically digested sludge. • Dewatered sludge produced from method is easier to remove compared to sludge produced from sand drying bed.
Disadvantages
• High capital cost compared to other types of drying bed.

sludge to float without causing upward or downward pressure across the wedge wire surface.

- After the proper amount of sludge has been introduced, the initial separate water layer and drainage water are allowed to percolate away at a controlled rate through the outlet valve.
- After the controlled drainage phase, the sludge is allowed to further dewater by natural drainage and then it can be removed.

Wedge wire drying bed has several advantages and disadvantages. These are presented in Table 47.

3.4.3.4. Vacuum-assisted drying beds

Vacuum-assisted drying beds combines several features of both sand drying beds and mechanical dewatering systems.[89] Historically, vacuum-assisted drying beds have been built in 1976 at Sunrise City, Florida. The dimension of particular operating unit is 20 ft × 40 ft (6 m × 12 m).[5] This system comprises of a reinforced concrete, a layer of aggregate (several inches) and a rigid multi-media on top where the sludge will be placed on. Vacuum chamber is placed between the reinforced concrete slab and the multi-media layer. A vacuum pump is then connected to draw vacuum from the sump. Basically, the operating sequences for vacuum-assisted drying bed are as follows[5,23]:

- When the conditioned sludge is introduced to the bed surface, dewatering process begins by gravity flow.
- When the maximum sludge level in the bed reaches 12–18 in. (30–46 cm), the flow of conditioned sludge is stopped. The operation of vacuum pump begins at 1–10 in. (2.5–25 cm) of mercury or at pressures between 3 and 34 kN/m^2.

Table 48. The Advantages and Disadvantages of Using Vacuum-assisted Drying Bed.[88,89]

Advantages
Requires small land area than sand drying beds.
Greater operational control than sand drying beds.
Requires lower costs than mechanical dewatering systems.
Ease at operation compared with mechanical dewatering systems.
Has short cycle time.
Disadvantages
Labor intensive.
Expensive to operate than sand drying beds.

- The vacuum pump is shut off when the sludge cake formed on the bed cracks. Dewatered sludge is ready to be removed by using a front-end loader.

After the process has finished, the remaining sludge residue is removed by washing the multi-media layer with a high-pressure hose. This can be done before they are used again for another application. Table 48 presents several advantages and disadvantages of using vacuum-assisted drying bed.

3.4.4. *Mechanical Sludge Dewatering Systems*

Mechanical sludge dewatering systems include belt filter presses, recessed-plate filter presses, vacuum filters, and centrifuges.[1] Generally, belt filter presses, recessed-plate filter presses, vacuum filters, and centrifuges have a great potential to achieve 20–32%, 35–45%, 12–22%, and 25–35% of solids content, respectively.[2]

3.4.4.1. Vacuum filters

Vacuum filters are the oldest mechanical sludge dewatering systems that have been used widely in most of the WWTPs. In 1872, vacuum filters were patented in England by William and James Hart. The application of this system in the United States begun in the mid-1920s where the vacuum filters are used to dewater municipal wastewater sludge.[5] Until the 1960s, the drum or scraper-type rotary vacuum filter was predominant. After that, in the 1970s, the belt-type filter with natural or synthetic fiber cloth, woven stainless steel mesh, or coil spring media has become dominant and widely used in full-scale operation. A vacuum filter comprises of a horizontal cylindrical drum that rotates partially submerged in a vat of sludge. The drum surface is divided into a number of compartments that are connected to a rotary valve. Bridge blocks in the valve divide the drum compartments into three zones, which are cake

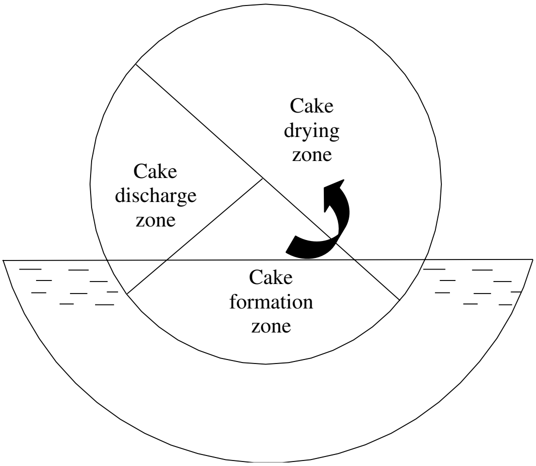


Figure 24. Operating Zones of a Rotary Vacuum Filter.[5]

formation zone, cake drying zone, and cake discharge zone. Figure 24 illustrates the operating zones of a rotary vacuum filter.

About 25% (normally around 10–40%) of the drum surface is submerged in a vat of conditioned sludge. The portion where the drum submerged in the sludge is referred to as the cake formation zone. Vacuum applied to the submerged drum section causes filtrate to pass through the media and sludge cake to be retained on the media. As the drum rotates, each section is carried through the cake formation zone to the cake drying zone. Normally, the cake drying zone represents from 40% to 60% of the drum surface and ends at the point where the internal vacuum is shut off. At this position, the sludge cake and drum section enter the cake discharge zone, where the sludge cake is removed from the media.[23]

It has been mentioned before that until the 1960s, the drum or scraper-type rotary vacuum filter was predominant. Since then, the belt-type filter has become dominant and widely used in full-scale operation. There are two coverings that are most commonly used with belt type units, which are coil springs and fiber cloths. This unit differs from the drum or scraper units because the drum covering leaves the drum. This vacuum filter type uses two layers of stainless steel coils arranged around the drum. The two layers of springs leave the drum and are separated from each other after the dewatering process. Through the separation of these layers, the cake is lifted off the lower layer of springs and can be discharged from the upper layer. Cake release is usually not a problem if the sludge is properly conditioned. After the cake is discharged, the coil springs are washed and returned to the drum.[5,23]

Another type of coverings used with belt type filter is fiber cloths. Usually fiber cloths are required when filtering unthickened sludge. The filter media in this system leaves the drum surface at the end of the drying zone and passes over a small diameter discharge roll and before it returns to the drum for another cycle. Normally, fiber cloth filter has a small diameter curved bar between the point where the belt leaves the drum and the discharge roll. Cleaner filtrate will be produced from cloth media made of staple fiber compared to cloth media made from monofilament fiber. Basically, the performance of vacuum filters is influenced by several criteria such as yield, the efficiency of solids removal, and the cake characteristics.

Table 49 illustrates some of the advantages and disadvantages of vacuum filtration compared to other dewatering processes. Meanwhile, Table 50 lists the common design shortcomings of vacuum filters.

Table 49. The Advantages and Disadvantages of Vacuum Filtration.[23]

Advantages

- Ease of operation (formation and discharge of sludge cake are easily visible).
- Continuous operation.
- The system will continue to operate even if the dosage of chemical conditioner is not optimized.
- Low maintenance requirements.
- Coil spring media can sustain for a long time compared to cloth filter media.

Disadvantages

- Use large amount of energy.
- Create noise pollution due to the application of vacuum pump.
- Production of ammonia odors due to the application of lime for conditioning.
- Cause maintenance and cleaning problems due to the application of lime and ferric chloride for conditioning.
- Potentially to emit strong odors if the sludge is poorly stabilized.

Table 50. Common Design Shortcomings of Vacuum Filter Installations.[23]

Shortcomings	Resultant Problems	Solution
Improper filter media	• Filter blinds. • Provide in adequate solids capture. • Produce poor sludge cake.	Replace media after testing for optimum media.
Improper chemical conditioner used	• Poor solid capture. • Low solid loading rate. • Low cake solid concentration.	Replace with correct chemical conditioners.
Inadequate water pressure for spray nozzles	Improper cleaned media.	Proper booster pumping to maintain 345 kPa minimum pressure.

3.4.4.2. Belt filter presses

Belt filter presses have been used in the United States in the 1960s. This system was developed by Klein and by Smith and Loveless. This method is a common type of mechanical dewatering equipment and can be applied to dewater most of the sludge generated at municipal WWTPs.[90] Belt filter presses use single or double moving belts to continuously dewater sludge through or more stages of dewatering. Basically, the general mechanical components of a belt filter press are dewatering belts, rollers and bearings, belt tracking and tensioning systems, control and drives, and a belt washing system.[90] Most of the belt filtration processes include three basic operational stages, which are chemical conditioning, gravity drainage, and shear and compression dewatering stages.[5,23,91] Figure 25 illustrates a simple belt filter press and shows the location of the three stages.

Proper chemical conditioning is very important for successful and consistent performance of the belt filter press. An organic polymer (flocculant) is added to the sludge before its being fed to the belt press. Free water drains from the conditioned sludge in the gravity drainage stage of the press. Typically, it takes about 1 or 2 min for drainage. The sludge then enters a two-belt contact zone, where a second upper belt is gently set on the forming sludge cake. The belts with the capture cake between them pass through rollers, which have small diameter. This stage subjects the sludge to continuously increasing pressures and shear forces. At this stage, more water is expelled throughout the roller section to the end where the sludge cake is discharged. In order to remove the sludge cake from the belts, a scraper blade is often applied for each belt at the discharge section. Meanwhile, the wash spray is generally provided for belts cleaning purposes.

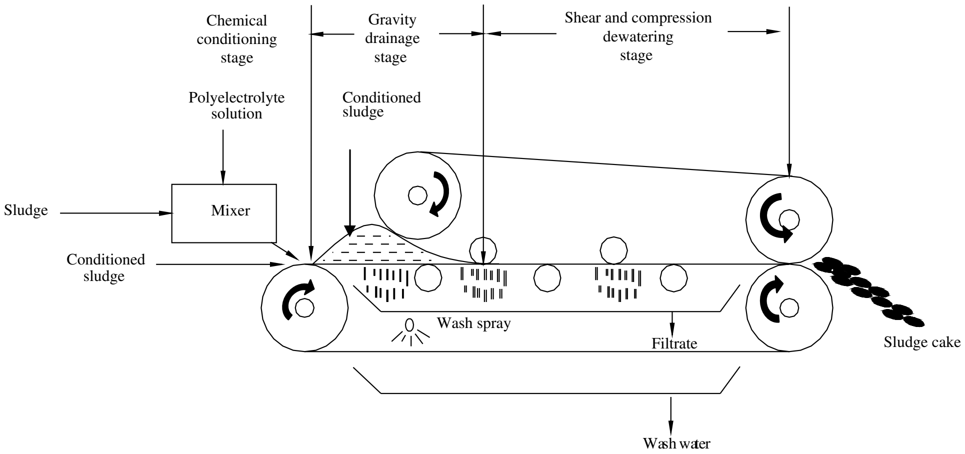


Figure 25. The Three Basic Stages of a Belt Filter Press.[5,23,91]

Table 51. The Advantages and Disadvantages of Belt Filter Press.[5,23,90]

Advantages

- High pressure machines are capable of producing drier cake than any machine.
- Low staffing requirements.
- Low power requirements.
- Less noise and vibration associated with belt press compared to centrifuges.
- Relatively simple in maintenance.
- The system can be started and shut down quickly compared to other types of dewatering unit such as centrifuges.
- Easy for the inexperienced worker to understand the operation.
- Continuous operation.

Disadvantages

- Very sensitive to incoming feed characteristics and chemical conditioning.
- Has a potential to emit noticeable odors if the sludge is poorly stabilized.
- Require greater operator attention than centrifuge.
- Require prescreening or grinding to minimize the risk of sharp objects damaging the belt.
- Require higher polymer dosage than a centrifuge.
- Higher concentration of oil and grease in the sludge can result in blinding the belt filter lower solids content cake.
- Short media life as compared to other devices using cloth media.

Usually, the performance of belt filter press is measured by the percentage solids of the sludge cake, the percentage solids capture, the solids and hydraulic loading rates, and the required polymer (chemical conditioner) dosages.[23] Table 51 illustrates some of the advantages and disadvantages of the belt filter press compared to other dewatering processes. Meanwhile common design shortcomings associated with belt filter press installations are listed in Table 52.

3.4.4.3. Recessed-plate filter presses

Recessed-plate filter presses are among the oldest types of mechanical dewatering devices. These devices have great potential to produce the highest cake solids concentration of any mechanical dewatering equipment. They are commonly applied in industrial applications than in municipal wastewater facilities.[92] Historically, in the early 1920s, the first U.S. municipal sludge dewatering installations were located in Worcester, Massachusetts and Providence, Rhode Island.

The two types of filter presses that are commonly available to dewater municipal wastewater sludges are fixed volume recessed-plate filter press and the variable volume recessed-plate filter press (diaphragm filter press).[5,23,92] The fixed volume recessed-plate filter press comprises of a series of parallel plates, each is fitted with a filter cloth and held together in a rigid framework. In this device, conditioned sludge is pumped into a volume between two filter cloths that held in place by a

Table 52. Common Design Shortcomings of Vacuum Filter Installations.[23]

Shortcomings	Resultant Problems	Solution
Improper belt type	Frequent tearing or wrinkling or inadequate solids capture.	Experiment with different belt types and install proper belt for actual conditions.
Improper tracking of filter belt	Belt creeps off rollers and dewatering operation must be stopped for repair.	Repair or adjust automatic tracking device, if one exist. If not, add such a device.
Inadequate control of conditioning	Frequent underconditioning or overconditioning of sludge.	Install a feedback control system that monitors sludge solid content and sets required polymer addition.
Wash spray unit poorly sealed	Fine mist escapes from spray wash unit increasing moisture/corrosion problems.	Replace or modify wash spray unit to provide better seal around belt.
Inadequate mixing time for polymer and feed sludge before belt press	Underconditioning of sludge.	Move polymer injection point upstream toward feed pumps to increase mixing time or install polymer/sludge mixing before belt presses.
Inadequate wash water supply	Sludge formed on belts and/or rollers.	Increase spray water pressure or install new spray heads.

rigid framework. As a result of high pressure, a large portion of water in the feed sludge passes through the filter cloth and drains from the filter press. Sludge solids and the remaining water finally fill the void volume between the filter cloths and continued pumping of solids to the filter press is no longer productive. At this point, pumping is stopped and the filter press is opened to release the dewatered sludge onto a conveyor belt for removal before starting new cycle. Usually, cake breakers are required to break up the rigid cake into conveyable form. In this method, filter media is used on both sides of the filtering volume. Figure 26 illustrates cross section of a fixed volume recessed-plate filter press assembly.

The construction of a variable volume recessed-plate pressure filter press is similar to the fixed volume recessed-plate filter press except that a rubber diaphragm is placed behind the media. A dewatering process begins in this device when the conditioned sludge is introduced into each chamber from a slurry inlet pipe, which is located at the top or bottom of each plate. Usually, it takes about 10–20 min to fill the filter press and reach an end point.[4] The sludge feed pump is automatically turned off when the end point is reached. Water or air that is under high pressure is then pumped into the space between the rubber diaphragm and plate squeezing the already formed and partially dewatered cake. Typically, 15–30 min of constant pressure are required to dewater the cake to the desired solids content.[4] At the end

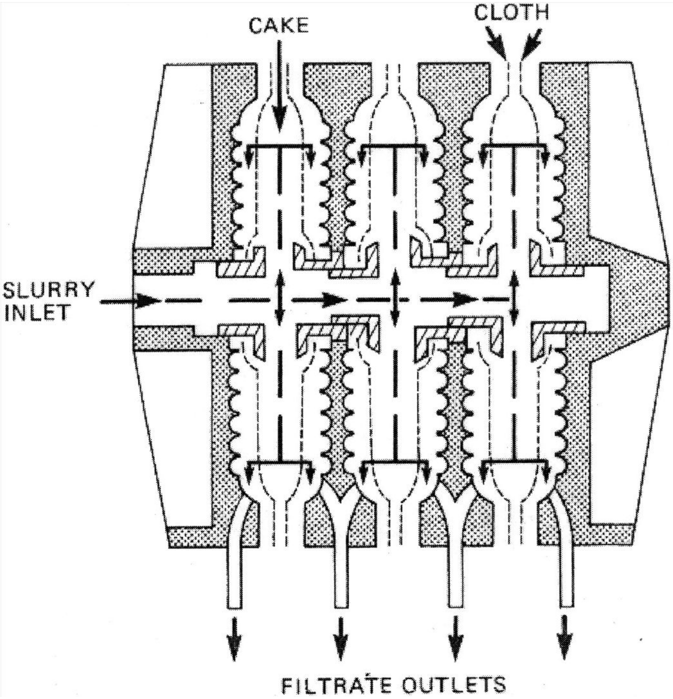


Figure 26. Cross Section of a Fixed Volume Recessed-Plate Filter Assembly.[5,23,93]

of the process, water is returned to a reservoir, plates are automatically opened, and sludge cake is removed. Figure 27 shows the cross section of a variable volume recessed-plate filter assembly.

Table 53 lists the advantages and disadvantages of recessed-plate filter presses compared to other dewatering processes. Meanwhile common design shortcomings associated with recessed-plate filter presses installations are listed in Table 54.

4. Land Applications of Biosolids

4.1. *Introduction*

Land application is the application of biosolids to the land either to condition the soil or to fertilize crops or other vegetation grown in the soil. Nearly half of the biosolids production in the United States is currently being used beneficially to improve soils. It is well suited for managing solids from any size wastewater treatment facility. As the method of choice for small facilities, it offers cost advantages, benefits to the environment, and value to the agricultural community.

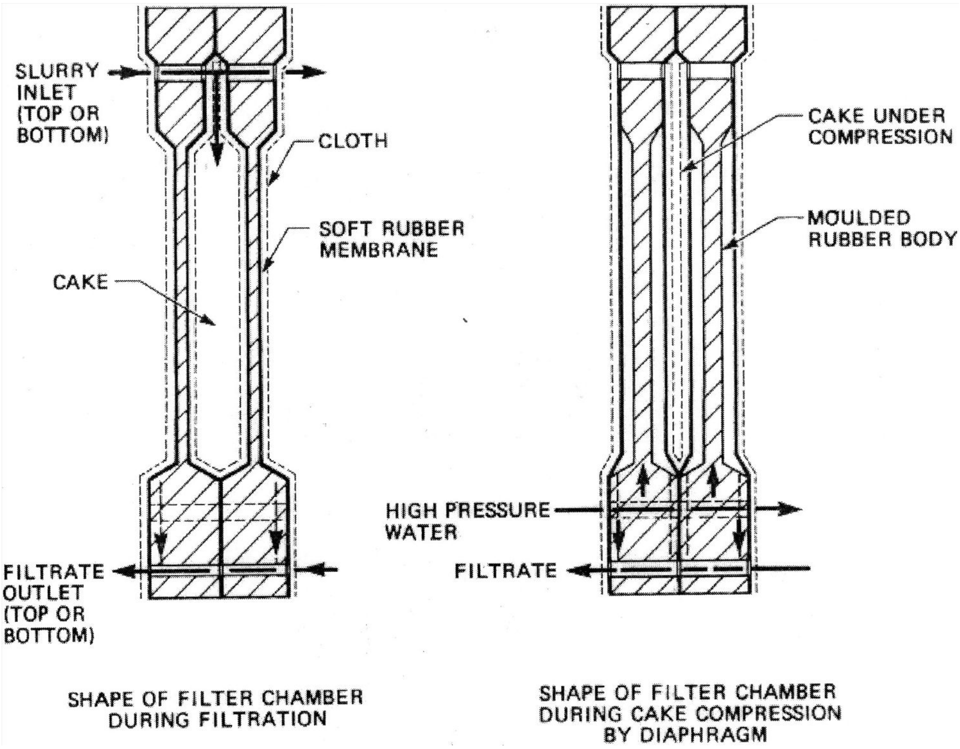


Figure 27. Cross Section of a Variable Volume Recessed-Plate Filter Assembly.[5,23]

Table 53. The Advantages and Disadvantages of Recessed-Plate Filter Presses.[5,23,92]

Advantages

- Highest cake solids concentration.
- Very high solid capture.
- Little or no operator attention during dewatering phase.
- The application of lime as a conditioner helps to stabilize and disinfect the final product.
- Produces a drier cake for diaphragm filter press.
- Easier to dose polymer for diaphragm filter press.

Disadvantages

- The process is mechanically complex.
- High capital costs.
- Requires large land area.
- Requires special support structure.
- High labor costs.
- Batch operation produces more heterogenous influent.
- Requires grinder or prescreening equipment on the feed.

Table 54. Common Design Shortcomings of Recessed-Plate Filter Presses Installations.[23]

Shortcomings	Resultant Problems	Solution
Insufficient filter cloth washing	Blinding of filter cloths, poor cake release, longer cycle time required, and wetter cake.	Increase the frequency of washing.
Improper conditioning chemicals utilized	Blinding of filter cloths or poor cake release.	Switch conditioning chemicals or dosages.
Improper filter cloth media specified	Poor cake discharge and difficult to clean.	Change media.
Feed sludge is too dilute for efficient filter press operation	Long cycle time and reduced capacity.	Thicken sludge before feeding to filter press.
Sludge feed at only one end of large filter press	Unequal sludge distribution within the press.	Use equalization tank or centrifugal pump to feed at opposite end of press.

Over the past several years, the quality of municipal sludges has improved dramatically due in part to enforcement of federal, state, and local regulations and in part to the pretreatment standards that indirect dischargers, such as industries, must comply with before they send their wastewater to public facilities for final treatment. The notion of nourishing the land with human animal wastes is not new. For thousands of years, farmers have recognized the value of human manure as a fertilizer. In fact, sewage sludge has been applied to the land in the United States and Europe for over 40 years. Today, over half of the sewage sludge generated in the United States is land applied.

Recycling biosolids through land application serves several purposes. It improves soil properties, such as texture and water holding capacity, which make conditions more favorable for root growth and increases the drought tolerance of vegetation. Biosolids application also supplies nutrients essential for plant growth, including nitrogen and phosphorous, as well as some essential micronutrients such as nickel, zinc, and copper.

Biosolids can also serve as an alternative or substitute for expensive chemical fertilizers. The nutrients in the biosolids offer several advantages over those in inorganic fertilizers because they are organic and are released slowly to the growing plants. These organic forms of nutrients are less water soluble and, therefore, less likely to leach into groundwater or run off into surface waters.[94]

Land application is most easily implemented where agricultural land is available near the site of biosolids production, but advances in transportation have made land application viable even where hauling distances are greater than 1,000 miles. Biosolids are generally land applied using one of several techniques. The biosolids may be sprayed or spread on the soil surfaces and left on the surface. They may be tilled (incorporated) into the soil after being surface applied or injected directly below the surface for producing row crops or other vegetation and for establishing lawns. Biosolids in a liquid state can be applied using tractors, tank wagons, irrigation systems, or special application vehicles. Dewatered biosolids are typically applied to land using similar equipment to that used for applying limestone, animal manure, or commercial fertilizer. Both liquid and dewatered biosolids are applied to land with or without subsequent incorporation into the soil.[95]

4.2. *The Risk of Applying Biosolids to the Land, Health, and Environment*

Biosolids are a complex mixture that can contain pollutants from household, commercial, and industrial wastewaters such as metals, pathogens (which are disease-causing organisms), chemical pollutants such as medicines, and synthetic organic compounds such as polychlorinated biphenyls (PCB). At uncontrolled levels, these pollutants could accumulate in soils or crops, which could potentially affect our health and the health of animals, as well as the longstanding health of our lands and waters. Poor management of biosolids could become harmful to public health and to the environment.

Infection by microorganisms is a major risk from human contact with partly treated products derived from wastewater depending on the treatment process and source, biosolids may contain bacteria, viruses, helminthes (such as hookworm), protozoans (such as *Giardia*), and fungi. There are a number of recognized exposure pathways including inhalation, ingestion (especially through hand to mouth contact), and contact with broken skin. Therefore, all persons handling biosolids need to observe personal hygiene precautions. However, pollutant levels can be controlled and one must recognize that fertilizers and pesticides also pose similar types of risks.[96]

Due to the diverse sources of wastewater, biosolids may also contain significant concentrations of chemical contaminants, including heavy metals and pesticides. Chemicals present in biosolids may become dissolved and move through soils. If biosolids are used for agricultural purposes, this may allow chemicals to pass into the food chain, which could have both health and economic implications. As some chemical compounds may remain in the environment for long periods, the frequency of applications to one site needs to be managed to prevent contaminant levels being built up to harmful levels.

There are also environmental constraints that can limit the acceptability of biosolids application. Despite many positive impacts to the environment, land application can have negative impacts on water, soil, and air if not practiced correctly.

Issues for consideration include the potential to pollute surface water and groundwater with nutrients or trace contaminants, the importation of exotic plant seeds and spores to sensitive areas and alteration of the pH of some soils, which may increase the mobility of some metals.

Excess nutrients in the biosolids, which is primarily nitrogen compounds, can leach from the soil and reach groundwater. Runoff from rainfall may also carry excess nutrients to surface water. This can results a negative impact to water if the rates exceed the nutrient requirements of the vegetation. However, because biosolids are a slow-release fertilizer, the potential for nitrogen compounds to leach from biosolids-amended soil is less than that posed by the use of chemical fertilizers. In areas fertilized by either biosolids or chemicals, these potential impacts are mitigated by proper management practices, including the application of biosolids at agronomic rates (the rate nutrients are used by the vegetation). Maintenance of buffer zones between application areas and surface water bodies and soil conservation practices will minimize impacts to surface water.

Primary negative impact to the air is the odors from biosolids application. It is a greater nuisance rather than threat to human health or environment. Odor controls focus on reducing the odor potential of the biosolids or incorporating them into the soil. Stabilization processes such as digestion can decrease the potential for odor generation. Biosolids that have been disinfected through the addition of lime may emit ammonia odors but they are generally localized and dissipate rapidly. The stabilization of biosolids reduces odors and usually results in an operation that is less offensive than manure application.

Mismanagement of a biosolids land application can result in negative impacts to soil. Federal regulations contain standards related to all metals of concern. Application of biosolids should comply with these standard to ensure that the accumulation of metal not reach harmful levels. Stringent record keeping and reporting requirements on both the federal and state levels are imposed to prevent mismanagement.

These guidelines provide information on acceptable uses of biosolids. They have been developed to facilitate responsible and beneficial reuse and minimize the risk of any adverse effects of biosolids application to human health, animal health, and the environment.

Generally, we can manage these risks by:

- Promoting proper pollutant source control and disposal of household and business hazardous wastes.
- Assessing biosolids quality.
- Assuring appropriate land types and use for application while verifying compatibility with surrounding areas.

- Determining appropriate soil, landscape, and crop or vegetative conditions for biosolids use or restriction.
- Monitoring and overseeing transport, storage, application, and land use during and after application.
- Limiting harvest or grazing until appropriate periods has elapsed. These components are all included in current federal and state regulations. Additional protective measures, such as local government and public notification, will be refined through amendments to the state regulations.

4.3. *Advantages and Disadvantages*

There are some advantages as well as several disadvantages offered by land application of biosolids that must be considered before selecting this option for managing this material.

4.3.1. *Advantages*

Land application is an excellent way to recycle wastewater solids as long as the material is quality controlled. It returns valuable nutrients to the soil and enhances conditions for vegetative growth. The nitrogen in this organic fertilizer is primarily organic N, which is released slowly over time; therefore, it is available when the crop needs it and minimizes potential for nitrate leaching. Biosolids also have many other essential elements needed for plant growth. Farmers from various parts of the world use it for its phosphorus and zinc because these two elements are deficient in the soils of various regions. It is especially beneficial when applied to soils that have been subject to soil erosion, excavated, or have phosphorous deficiencies. The biosolids improve soil water infiltration, which helps minimize soil erosion.

Land application is a relatively inexpensive option and capital investments are generally lower than other biosolids management technologies. Contractors can provide the necessary hauling and land application equipment. In addition, on-site spatial needs can be relatively minor depending on the method of stabilization selected.

4.3.2. *Disadvantages*

The process of land application can be labor intensive although it requires relatively less capital. Even if contractors are used for application, management oversight is essential for program success. Land application also is a potential public opposition, which is encountered most often when the beneficial use site is close to residential areas. Odor problem is the primary reason of public concern. In worst case, municipalities or counties may pass ordinances, which ban or restrict the use of biosolids. However, many successful programs have gained public support through effective communications, an absolutely essential component in the beneficial use of biosolids.

Another disadvantage of land application is limited to certain times of the year, especially in colder times. This is because biosolids cannot be applied to frozen or snow-covered grounds, while farm fields are sometimes not accessible during the growing season. Hence, it is often necessary to provide a storage capacity in conjunction with land application programs. Although the timing is right, weather can interfere with the application. Spring rains can make it impossible to get application equipment into farm fields, making it necessary to store biosolids until weather conditions improve.

4.4. *Regulation and Standards*

In 1993, the USEPA put into affect Title 40 of the CFR, Part 503, *The Standards for the Use or Disposal of Sewage Sludge.* This regulation was developed as a requirement by the Clean Water Act Amendments of 1987, to protect public health and the environment from any reasonably anticipated adverse effects of certain pollutants that might be present in sewage sludge.[97] The USEPA's 40 CFR Part 503, *Standards for the Use and Disposal of Sewage Sludge* (5), requires that wastewater solids be processed before they are land applied. The Part 503 rule established a set of requirements for the three most common final uses or disposal methods of biosolids generates at a WWTP. These three uses or disposal methods include land application as a soil conditioner or fertilizer, surface disposal at a specific final disposal site, and incineration. 40 CFR Part 503 represents a minimum standard which, according to current scientific information, will allow safe use of the nutrients and beneficial organic properties of biosolids. Table 55 lists the regulations and exclusions in the new rule.

Land application of biosolids to condition or fertilize soil for crops or other vegetation is the most common beneficial use of biosolids in the United States. Some examples of land that benefits from biosolids application and are not frequently visited by the public include agricultural land, forest land, and reclamation sites.

Table 55. Sludge Types and Practices Regulated by 40 CFR Part 503.[98]

Regulated	Excluded
Land application systems	Processing prior to use or disposal
Surface disposal system	Industrial or hazardous sludges
Monofills (sludge-only landfills)	Sludge with $>$ 50 ppm PCB's
Sludge incineration systems	Co-disposal or co-firing with municipal solid waste

Some examples of land where the public is likely to come into contact with biosolids include public parks, plants nurseries, roadsides, golf course, lawns, and home gardens. Depending upon the land application use, three EPA regulations must be addressed, which are pollutant concentration (PC) limit for heavy metals, Class A or Class B pathogen reduction, and vector attraction reduction. Record keeping and reporting requirements are also required by the EPA in conjunction with monitoring and are detailed in Part 503.[98]

4.4.1. *Options for Meeting Land Applications Requirements*

The EPA has established four options for meeting land application requirements pertaining to pollutant limits, pathogen class, and vector attraction reduction. Each of these four different options has various land application alternatives.[33] The options are:

i. The exceptional quality (EQ) option,
ii. The PC option,
iii. The cumulative pollutant loading rate (CPLR) option, and
iv. The annual pollutant loading rate (APLR) option.

4.4.1.1. The EQ option

Biosolids meet a low pollutant and Class A pathogen reduction, and can be used in bulk or given or sold in containers for unrestricted use where the general public may come into contact with the biosolids.

4.4.1.2. The PC option

The PC are used along with the pathogen level and requirements to stabilize organic matter as quality standards for EQ biosolids. Biosolids with metal concentrations above the PC require a permit for each site and the rate applied is based on the nitrogen need of the crop on the site. The cumulative amount of metals must be tracked. PC biosolids may only be applied to land in bulk and are subject to management practices.

4.4.1.3. The CPLR option

This is the maximum amount of a pollutant that can be applied to a site over its lifetime by all biosolid applications meeting ceiling concentration limits. Biosolid applications must be discontinued when any one of the pollutants reaches its maximum CPLR.

4.4.1.4. The APLR option

APLR biosolids are biosolids that are sold or given away in a bag or other containers for land application and exceed the pollutant limits for EQ biosolids but meet the ceiling concentrations limits.

4.4.2. *Pollutant Limits of Heavy Metals*

The presence of trace elements sometimes referred to as heavy metals, in sewage sludges stems from the contaminants discharged into the sewage system as well as the effectiveness of sewage treatment. All sewage sludges contain trace elements, but concentrations tend to be lower than they once were since government regulations have reduced the amount of metals discharged by industry into the sewage system. EPA regulations limit the concentrations of 10 heavy metals in land-applied sewage sludges. These metals include arsenic, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, selenium, and zinc. Table 56 shows the concentrations limits for the pollutants.

Table 56. Maximum Metal Concentrations.[33,98]

Metal	Ceiling Concentration Limits for All Biosolids Applied to Land (mg/kg)	CPLR Limits for CPLR Biosolids (kg/hectare)	PC Limits for EQ and PC Biosolids (mg/kg)	APLR Limits for APLR Biosolids (kg/hectare per 365-day Period)
Arsenic	75	41	41	2.0
Cadmium	85	39	39	1.9
Copper	4,300	1,500	1,500	150
Lead	840	300	300	75
Mercury	57	17	17	15
Molybdenum	75	NL	NL	0.85
Nickel	420	420	420	—
Selenium	100	100	100	21
Zinc	7,500	2,800	2,800	5.0
Applies to:	All biosolids that are land applies	Bulk biosolids and bagged biosolids[c]	Bulk biosolids	Bagged biosolids[c]

[a] Dry-weight basis.

[b] As a result of February 25, 1994, Amendment to the rule, the limits for molybdenum were deleted from the Part 503 pending EPA reconsideration.

[c] Bagged biosolids are sold or given away in a bag or other container.

NL = No limit.

4.4.3. *Pathogen Class for Land Application*

Pathogens are discharged into sewers by infected individuals. Most are destroyed during wastewater treatment, but sufficient numbers remain in sewage sludge to infect people who work with or eat product from sludge-amended fields. The EPA has set requirements for land application of biosolids that depend on the level of pathogen reduction achieved at the WWTP. Based on the degree or amount of pathogen reduction, biosolids are categorized as Class A or Class B. Generally, if pathogens have been reduced below detectable levels, the biosolids meets Class A designation. If pathogens have been reduced to levels that do not pose a threat to public health or environment and actions are taken to prevent exposure to the biosolids after disposal, it meets Class B designation.[33]

4.4.3.1. Class A biosolids

Class A biosolids are essentially pathogen-free with no pathogen-related restrictions for land application. They usually are sold or distributed in urban areas for gardening, landscaping, or turf fertilization. The goal of requirements of Class A biosolids is to reduce the pathogens to below detectable levels. Class A biosolids can be land applied to areas where public contact is likely such as parks, lawns, gardens, and golf course and can be used as a soil conditioner or fertilizer where food crops are grown. The EPA has established six treatment alternatives for achieving Class A-level biosolids with respect to pathogen. Table 57 shows the treatment that approved by EPA.

The treatment alternatives listed in Table 57, involve technologies such as composting, heat drying, thermophilic aerobic digestion, beta-ray irradiation, gamma-ray irradiation, and pasteurization. The EPA has specific time and temperature criteria for each treatment alternative that must be met for a Class A level to be achieved. For clarification, because of the extensive treatment process and quality of the biosolids, EPA does not consider Class A biosolids to be generated waste, wastewater, or wastewater sludge that would require an approval in accordance with Section 23 of the *Activities Designation Regulations*. Upon the time Class A biosolids are used, sold, or disposed of, the EPA requires that one of the following criteria must be met by any of the approved treatment processes:

- The density of fecal coliform in the biosolids must be less than 1000 most probable numbers (MPN) per gram total solids (dry-weight basis).
- The density of *Salmonella* sp. in the biosolids must be less than 3 MPN per grams of total solids (dry-weight basis).

4.4.3.2. Class B biosolids

Class B biosolids are not treated and stabilized to the same extent as the Class A biosolid. These municipal biosolids meet a lower quality standard for metal,

Table 57. EPA-Approved Treatment Alternatives for Meeting Class A Pathogen Requirements.[98]

No.	Treatment Alternatives	Treatment Description
1	Thermally treated biosolids	Biosolids must be subjected to one of four time-temperature regimes.
2	Biosolids treated in a high pH–high temperature process	Biosolids must meet specific pH, temperature, and air-drying requirements.
3	Biosolids treated in other process	Demonstrate that the process can reduce enteric viruses and viable helminth ova. Maintain operation conditions used in the demonstration after pathogen reduction demonstration is completed.
4	Biosolids treated in unknown process	Biosolids must be tested for pathogens — *Salmonella* sp. or fecal coliform bacteria, enteric viruses, and viable helminth ova — at the time the biosolids are used or disposed, or, in certain situations, prepared for used or disposal.
5	Biosolids treated in a process to further reduce pathogens (PFRP)	Biosolids must be treated in one of the PFRPs.
6	Biosolids treated in a process equivalent to a PFRP	Biosolids must be treated in a process equivalent to one of the PRFPs, as determined by the permitting authority.

pathogen, and contaminant concentrations. Due to the composition of Class B biosolids, EPA considers a Class B municipal biosolid as a generated waste that requires an approval in accordance with Section 23 of the *Activities Designation Regulations*. Class B biosolids have less stringent standards for treatment than Class A biosolids. Site restrictions for land application specific to crop harvesting, animal grazing, and potential public contact are required through regulations. The regulations for Class B biosolids are shown in Table 58. Similar to EPA-approved Class A treatment technologies; EPA has established three treatment alternatives for achieving Class B-level biosolids with respect to pathogens. Table 59 shows the treatment alternatives for meeting Class B pathogen requirements.

These treatment technologies include aerobic digestion, air drying, AD, composting, and lime stabilization. The EPA has specific time and temperature criteria for each treatment alternative must be met for a Class B level to be achieved. If the time and temperature are met for the specific treatment alternative, it is assumed that a Class B level was achieved. The values are listed in Table 60.

4.4.4. *Vector Attraction Reduction for Land Application*

Vectors are organisms, such as flies, that can carry pathogens from one location, such as a sludge-amended field, to another where infection can take place, such as Sunday picnic spot. The most cost-effective way to diminish this threat is to reduce

Table 58. EPA Site Restriction for Class B Biosolids Applied to Land.[99]

Potential Site	Site Restrictions
Food crops with harvested parts that touch the biosolids/soil mixture	Food crops with harvested parts that touch the biosolids/soil mixture and are totally above the land surface shall not be harvested for 1–4 months after application of biosolids.
Food crops with harvested parts below the land surface	Food crops with harvested parts below the surface of the land shall not be harvested for 20 months after the application of biosolids when the biosolids remain on the land surface for 4 months or longer prior to incorporation into the soil. Food crops with harvested parts below the surface of the land shall not be harvested for 38 months after the application of biosolids when the biosolids remain on the land surface less than 4 months prior to incorporation into the soil.
Food crops with harvested parts that do not touch the biosolids/soil mixture, feed crops, and fiber crops	Food crops with harvested parts that do not touch the biosolids/soil mixture, feed crops, and fiber crops shall not be harvested for 30 days after application of biosolids.
Animal grazing	Animal should not be grazed on the land for 30 days after application of biosolids.
Turf growing	Turf growing on land where biosolids are applied shall not be harvested for 1 year after application of the biosolids when the harvested turf is placed on either land with a high potential for public exposure or a lawn, unless otherwise specified by the permitting authority.
Public access	Public access to land with high potential for public exposure shall be restricted for 1 year after application of biosolids. Public access to land with a low potential for public exposure shall be restricted for 30 days after application of biosolids.

Table 59. EPA-Approved Treatment Alternatives for Meeting Class B Pathogen Requirements.[98]

No.	Treatment Alternatives	Treatment Description
1	The monitoring of indicator organisms	Test for fecal coliform density as a factor for all pathogens. The geometric mean of seven samples shall be less than 2 million MPN per gram per total solids or less than 2 million CFU per gram of total solids at the time of use or disposal.
2	Biosolids treated in processes to significantly reduce pathogen (PSRP)	Biosolids must be treated in one of the PSRPs.
3	Biosolids treated in a process equivalent to a PSRP	Biosolids must be treated in a process equivalent to one of the PSRPs, as determined by the permitting authority.

Table 60. PSRPs for Meeting Class B Pathogen Requirements.[98]

Process	Process Requirements
Aerobic digestion	Biosolids are agitated with air or oxygen to maintain aerobic conditions for a specific mean cell residence time at a specific temperature. Values for the mean cell residence time and temperature shall be between 40 days at 20°C and 60 days at 15°C.
Air drying	Biosolids are dried on sand beds or on paved or unpaved basins. The biosolids dry for a minimum of 3 months. During second or third month, the ambient average daily temperature is above 0°C.
Anaerobic digestion	Biosolids are treated in the absence of air for a specific mean cell residence time at specific temperature. Values for the mean cell residence time and temperature shall be between 15 days at 35–55°C and 60 days at 20°C.
Composting	Using the within-vessel, static aerated pile, or windrow composting methods, the temperature of the biosolids is raised to 40°C or higher and maintained for 5 days. For 4 h during the 5-day period, the temperature in the compost pile exceeds 55°C.
Lime stabilization	Sufficient lime is added to the biosolids to raise the pH of the biosolids to 12 after 2 h of contact.

the attractiveness of the applied sludge as a food source or breeding place. The EPA requires "vector attraction reduction" for all biosolids, whether they are Class A or Class B. The EPA describes 10 approaches for satisfying its vector attraction reduction requirement. 12 options to meet vector attraction reduction requirements when land applying biosolids are described in Part 503 and state Part 24 Rules.[99] These options can be grouped into two general approaches:

i. Preventing vectors from coming into contact with the biosolids by the use of physical techniques such as biosolids incorporation and biosolids injection below the soil surface within specified periods.
ii. Reducing the attractiveness of the biosolids to vectors by using specified organic matter stabilization processes such as composting, digestion, or adding alkaline/lime material.

Options to accomplish vector attraction reduction do not apply when biosolids or biosolids derivatives meet the criteria for EQ. EQ biosolids products are as safe as other agricultural and horticultural products and may be used without site restrictions. Depending upon the land application use or disposal method, one of the 12 treatment options is required. Table 61 shows the treatment options for reducing vector attraction.

Table 61. EPA-Approved Options for Meeting Vector Attraction Reduction.[98]

Option	Description of Option
1	Meet 38% reduction in volatile solids content.
2	Demonstrate vector attraction reduction with additional AD in a bench-scale unit.
3	Demonstrate vector attraction reduction with additional aerobic digestion in a bench-scale unit.
4	Meet a specific oxygen uptake rate for aerobically digested biosolids.
5	Use aerobic processes at greater than 40°C for 14 days longer.
6	Alkali addition under specified conditions.
7	Dry biosolids with stabilized solids to at least 75% solids.
8	Dry biosolids with unstabilized solids to at least 90% solids.
9	Inject biosolids beneath the soil surface.
10	Incorporate biosolids into the soil within 6 h of application to or placement on the land.
11	Cover biosolids places on a surface disposal site with soil or other material at the end of each operating day.
12	Alkaline treatment of domestic septage to pH 12 or above 30 min without adding more alkaline material.

4.5. *Design Criteria*

Design criteria for land application programs address issues related to application rates and suitable sites. Design criteria for physical facilities that are part of land application programs are discussed in separate fact sheets. Biosolids, site, and vegetative characteristics are the most important design factors to consider. Biosolids must meet regulatory requirements for stabilization and metals content. In addition, nutrient content and physical characteristics, such as percentage solids, are used to determine the appropriate application rate for the crop that will be grown and the soil in which the crops will be grown. Site suitability is determined based on factors such as soil characteristics, slope, depth to groundwater, and proximity to surface water.[100] In addition, many states have established site requirements further to protect water quality. Some examples include:

- Sufficient land to provide areas of nonapplication (buffers) around surface water bodies, wells, and wetlands.
- Depth from the soil surface to groundwater equal to at least one meter.
- Soil pH in the range of 5.5–7.5 to minimize metal leaching and maximize crop-growing conditions.

Site suitability is also influenced by the character of the surrounding area. While odors and truck traffic may not be objectionable in an agricultural area, both will adversely impact residential developments and community centers close to fields where biosolids are applied. The type of vegetation to be grown is also a design

Table 62. Typical Biosolids Application Scenarios.[95]

Type of Site/ Vegetation	Schedule	Application Frequency	Application Rate
Agricultural Land			
Corn	April, May, after harvest	Annually	5–10 dry tons per acre
Small grains	March–June, August, fall	Up to 3 times per year	2–5 dry tons per acre
Soybeans	April–June, fall	Annually	5–20 dry tons per acre
Hay	After each cutting	Up to 3 times per year	2–5 dry tons per acre
Forest land	Year round	Once every 2–5 years	5–100 dry tons per acre
Range land	Year round	Once every 1–2 years	2–60 dry tons per acre
Reclamation	Year round	Once	Sites 60–100 dry tons per acre

consideration. Vegetation, such as soil characteristics, will generally not exclude biosolids application since most vegetation will benefit from the practice. However, the type of vegetation will impact the choice of application equipment, the amount of biosolids to be applied, and the timing of applications. The effect of vegetation on the choice of application equipment is discussed above in the description of this technology. The amount of biosolids that may be applied to a site is a function of the amount of nutrients required by the vegetation and the amount of metals found in the biosolids. Table 62 summarizes the application frequency, timing, and rates for various types of sites.

Another factor to be considered in designing a land application program is the timing of applications. Long periods of saturated or frozen ground limit the opportunities for application. This is an important consideration in programs using agricultural lands; applications must be performed at times convenient to the farmer and must not interfere with the planting of crops. Most application of biosolids to agricultural land occurs in the early spring or late fall. As a result, storage or an alternate biosolids management option must be available to handle biosolids when application is not possible.

Forest lands and reclamation sites allow more leeway in the timing of applications. In some areas of the United States, application can proceed year round. Application is most beneficial on agricultural land in late fall or early spring before the crop is planted. Timing is less critical in forest applications when nutrients can be incorporated into the soil throughout the growing period. Winter application is less desirable in many locales. Rangelands and pasturelands also are more adaptable to applications during various seasons. Applications can be made as long as ground is not saturated or snow covered and whenever livestock can be grazed on alternate lands for at least 30 days after the application. The timing of single application in land reclamation programs is less critical and may be dictated by factors such as regulatory compliance schedules.[95]

4.6. *Site Selection and Management*

Once the biosolids grade and feasible end uses have been determined, the suitability of potential end use sites needs to be evaluated and appropriate management practices adopted.[101]

4.6.1. *Site Selection*

The initial phase of site selection and assessment should begin with a screening process. This activity should identify characteristics that will typically make a site unsuitable for biosolids application, unless the characteristics can be addressed through detailed management controls. For example, biosolids application would not typically be permitted on land that:

- Classified in a sensitive land category;
- Regularly subject to water logging or flooding; would be covered with snow at the time of application;
- Has surface rock outcrops over greater than 10% of the area (except for forestry and rehabilitation uses); or
- Has limited buffer distances.

Following these initial site considerations, characteristics such as soil structure, pH, and land slope and water table depth need to be evaluated as part of a land capability assessment.

Sites with only slight limitations should be able to receive regular biosolids applications without adverse effects linked to those characteristics. Site characteristics with more severe rankings should receive greater attention in the risk assessment and the scheme may need to be limited in terms of biosolids quality, application rate, and frequency. The interaction of parameters should be considered. As an example, moderate risks for soil acidity, soil permeability, and depth to groundwater could indicate a severe risk to groundwater from mobile metals.[101]

4.6.2. *Site Management Practices*

Following selection of potentially suitable end use sites, proponents will need to adopt effective site management controls to protect the environment, public health, and agriculture.[101] Appropriate management measures are required at all stages of biosolids use, including:

- site preparation,
- biosolids transport and storage,
- determining management controls; for example, application methods, signage, and fences around application sites,
- determining application rates, and
- monitoring post-application.

4.7. *Operation and Maintenance*

Land application systems generally use uncomplicated and reliable equipment. Operations include pathogen reduction processing, dewatering, loading of transport vehicles, transfer to application equipment, and the actual application. The other operations require labor skills of heavy equipment operators, equipment maintenance personnel, and field technicians for sampling, all normally associated with wastewater treatment facilities. In addition, the biosolids generator is responsible for complying with state and local requirements as well as federal regulations.

The biosolids manager must be able to calculate agronomic rates and comply with record keeping and recording requirements. In fact, the generator and land applier must sign certification statements verifying accuracy and compliance. The generator should also allocate time to communicate with farmers, landowners, and neighbors about the benefits of biosolids recycling. Control of odors, along with a viable monitoring program, is most important for public acceptance.[102]

4.8. *Cost*

It is difficult to estimate the cost of land application of biosolids without specific program details. For example, there is some economy of scale due to large equipment purchases. The same size machine might be needed for a program that manages 10 dry tons of biosolids per day as one managing 50 dry tons per day; the cost of that machine can be spread over the 10 or 50 dry tons, greatly affecting average costs per dry ton. One source identified costs for land application varying from \$60 to \$290 per dry ton. This range reflects the wide variety in land application methods as well as varying methods to prepare biosolids for land application. For example, costs for programs using dewatered biosolids include an additional step whereas costs for programs using liquid biosolids do not reflect the cost of dewatering. They do, however, include generally higher transportation costs.[102]

Despite the wide range of costs for land application programs, several elements must be considered in estimating the cost of any biosolids land application program:

- purchase of application equipment or contracting for application services,
- transportation,
- equipment maintenance and fuel,
- loading facilities,
- labor,
- capital, operation, and maintenance of stabilization facilities,
- ability to manage and control odors,
- dewatering (optional),
- storage or alternate management option for periods when application is not possible due to weather or climate,
- regulatory compliance, such as permit applications, site monitoring, and biosolids analyses, and
- public education and outreach efforts.

Land must also be secured. Some municipalities have purchased farms for land application; others apply biosolids to privately held land. Some operating costs can be offset through the sale of the biosolids material. Since the biosolids reduce the need for fertilizers and pH adjustment, farmers sometimes pay to have biosolids applied to their lands.

4.9. *Best Management Practices*

With increasing in production agriculture, many landowners are considering biosolids as a safe, inexpensive source of crop nutrients. Many landowners have found success using biosolids and eagerly recommend biosolids to their neighbors. The use of best management practices (BMPs) by WWTPs, haul truck operators, and landowners during the application of biosolids results in landowner satisfaction and good public perception. Unfortunately, not every WWTP is finding success with biosolids marketing and, all too often, practices have resulted in complaint generation. Just a little negative publicity can derail an application program. Consider the following factors as integral parts of a best management plan for land application of biosolids.

4.9.1. *Spills*

Any biosolids spilled onto highways must be cleaned up immediately. Some lime-stabilized biosolids are very slippery when wet, causing potentially hazardous conditions. Trucks hauling biosolids must be designed to prevent spillage onto roadways. Biosolids should not be loaded into dump trucks unless the truck bed is leakproof for the type of biosolids to be transported. Obviously, trucks must not be overloaded and transfer hoses must be completely emptied before entering roadways. It is mandatory that a proactive maintenance program on all biosolids application and hauling equipment are enacted and repairs are made before hazardous conditions result. A successful land application program will always pursue the goal of zero-complaint generation. Review protocol for spill cleanup and reporting with regulatory agencies before the accident happens.

4.9.2. *Odor*

All biosolids management plans must address odors as a potential problem. Nuisance complaints from odors are common and create an unfavorable public reaction. Potential for odors can be reduced by utilizing the following BMP:

- Incorporate or inject liquid biosolids soon after application to the site. Soil absorbs moisture from biosolids, which reduces odors as the biosolids dry. Faster drying or incorporation of biosolids at the application site will result in reduced odor generation.
- Avoid application to wet or waterlogged soil. Obviously, wet soils will absorb little water from applied biosolids, resulting in potential odors for longer periods. Similarly, biosolids stockpiled at the application site can produce odors until all are land applied. Minimizing the time biosolids are stockpiled can reduce odors and complaints.
- Use proper application rates. Over-applying biosolids can result in runoff and pools of liquid biosolids in low areas that can generate odors. Applying biosolids at an agronomic rate helps preventing these situations.
- Isolate application sites from residential, public access, and commercial areas. Keeping application sites away from these areas will limit potential for complaints.

4.9.3. *Safety Concerns*

Carefully evaluate all application sites, looking critically for any potential problems. For example, applying lime-stabilized biosolids near a school should not be a problem. However, if the biosolids dry and become dusty and if the wind direction changes and recess occurs while downwind of the application site, some children will probably experience burning eyes from the lime dust. Always evaluate the site and increase the border area around application sites near home sites, schools, or other public areas.

4.9.4. *Equipment Operation and Considerations*

All vehicles should be clean and routinely washed. Remember that the next load of material a dump truck will carry may not be biosolids. For example, a load of sand hauled behind a load of biosolids may smell like biosolids if the truck bed is not washed after the last load of biosolids. If biosolids are spilled onto the vehicle during loading, hose off the vehicle before hauling to the application site. Muddy conditions are often encountered at application sites. However, mud tracked onto roadways should be promptly removed to eliminate any hazardous conditions on roadways. Because people often smell what they see, odors can be reduced by using clean equipment. Again, keeping equipment clean, cleaning mud from roadways, preventing spills, etc., all enhance community acceptance and improve the public image of biosolids recycling. Consider the following BMP:

- Keep haul vehicles quiet and leave no mud on the road.
- Encourage equipment operators to keep all equipment clean. Daily cleaning of equipment, routine maintenance, and having conscientious equipment operators will go a long way toward improving public perception of recycling biosolids nutrients.
- Clean haul vehicle beds well before hauling other materials.

4.9.5. *Soils and Sites*

Carefully evaluate each site for potential groundwater and surface water contamination. Areas with karst topography require additional evaluation. Karst topography is underlain by limestone bedrock that has dissolved, resulting in landscapes with enclosed drainage systems. Over-application of biosolids in these areas may result in groundwater contamination.

At biosolids application sites, consider filter strips and borders around application sites as BMPs. Consider borders or filter strips in the following situations:

- Provide a minimum border of 100 ft around all application sites. No biosolids should be applied in these zones between application areas and roadways, streams, fencerows, etc.
- Provide a minimum border of 300 ft around all home sites, water wells, and sinkholes.
- For side-discharge spreaders, always throw biosolids toward the center of the field, never toward the outside.

Evaluating soils at the application site can help preventing ground and surface water contaminations. Soils that have either a high permeability (sandy soils) or a low permeability (clayey soils) may present special application planning. Evaluating soil texture in concert with slope and topography can help ensure that ground and surface waters are protected. Sites with slopes in excess of 8–12% should be avoided because of runoff. Likewise, soils shallow to bedrock provide little protection from groundwater contamination and should be avoided. The depth to groundwater at the site needs to be evaluated, when evaluating soils at application sites. Sites shallow to groundwater provide less buffer and greater potential for groundwater contamination. Following simple BMP for all biosolids applications is an effective way to recycle nutrients and build a more sustainable future.

4.9.6. *Haul Routes*

Haul routes and timing should be evaluated when considering application sites. Loud trucks roaring past school crossings or bus stops may cause some concerns from parents. Use of exhaust brakes, exceeding speed limits, and early morning and late evening hauling are all potential complaints. The BMP would be to avoid residential areas for all haul routes, especially before or after school. Trucks used for biosolids application are heavy and can cause considerable damage to roads if the same traffic patterns are used often. To avoid future problems, some dialogs with the local traffic department are needed before establishing the haul routes.

5. Use and Disposal of Biosolids

5.1. *Introduction*

Disposal of biosolids is a major environmental problem in the world. For many years, landfill and agricultural use were common disposal practices. However, regulatory considerations are becoming stronger in many parts of the world. Several strategies have been investigated with the goal of providing new options for biosolids treatment, use, and disposal. However, since these biosolids are rich in nutrients and organic matter, their utilization as soil conditioners appears to be one of the main sustainable options, especially for developing countries.

The US EPA in 1998 estimated that approximately 6,232,880 metric tons or 6,856,168 dry tons are generated annually in the United States.[103] Table 63 presents a breakdown of this estimate to show the quantities of biosolids estimated to have been generated by primary-only treatment publicly owned treatment works (POTWs) or secondary-or-above treatment POTWs and by privately or federally owned treatment works. Figure 28 shows the distributions for the most common methods of disposal

Table 63. Estimates of Biosolid Generation for Use or Disposal, 1998.[2]

Treatment Group	1998 Biosolid Generation (million U.S. dry tons)
Primary-only treatment POTW	0.5
Secondary-or-above treatment POTW	6.3
Privately and federally owned treatment works	0.1
Total	6.9

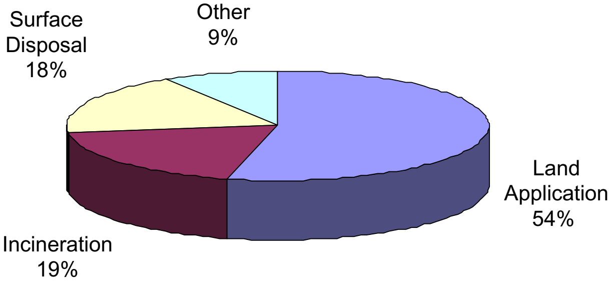


Figure 28. Use and Disposal of Sewage Sludge and Biosolids in the United States.[98]

or utilization in terms of percentage. Distribution and marketing, which include composting and heat drying, were estimated at approximately 3%. Ocean disposal has been discontinued since 1992. Because disposal to landfills has been severely curtailed in many states, land application has increased. Incineration is also decreasing because many incinerators cannot meet the Clean Air Act regulations or the US EPA 503 regulations.

In the United States, there are vast differences in the way biosolids is utilized or disposed. Table 64 shows the different disposal ways of biosolids. It shows that in New England and the Northeast, less than 30% is land applied. A significant amount of biosolids from the Northeast is shipped out of state for either land application or landfills. Nearly 90% of the biosolids produced in the Northwest are land applied.[2]

5.2. *Generation of Biosolids*

A variety of factors have influenced biosolids generation and use over the years. Advancements in wastewater and biosolids treatment technologies, including wastewater pretreatment, pollution prevention programs, and population growth, have resulted in increased volumes of higher-quality biosolids. Federal and state regulations and guidance, in particular the Part 503 Biosolids Rule, have encouraged recycling and use of biosolids rather than disposal. The U.S. population and the population served by municipal sewers have increased dramatically over the past 20 years. These increases have contributed to an increase in the volume of biosolids produced since 1972.

The total amount of biosolids produced annually has increased at a somewhat greater rate than the increased U.S. population being served by sewers. These increases in quantity have been accompanied by improved environmental quality of biosolids due to the greater prevalence of pretreatment and pollution prevention programs undertaken by industrial wastewater generators. The physical quality of biosolids also has improved because of advanced mechanical dewatering equipment, automated process control systems, aeration systems, and odor control systems. New

Table 64. Biosolids/Sewage Sludge Utilized or Disposed by US EPA Regions in the United States.[98]

Region	Population Served by POTWs[a]	Biosolids/ Sludge Production (dmt/yr)	Percentage Used/Disposed by			
			Land Application[b]	Surface Disposal[c]	Incineration	Other
I. New England	8,037,311	367,430	24	46	30	—
II. Northeast	21,726,101	605,046	30	14	23	33[d]
III. Mid Atlantic	18,152,556	1,040,206[e,f]	74	16	10	< 1
IV. Southeast	24,510,111	1,050,325[e]	57	30	12	1
V. North Central	35,587,804	1,705,316	51	2	30	17[g]
VI. South Central	21,150,172	425,203[e]	53	45	2	—
VII. Midwest	9,036,498	511,712[e]	65.5	4	25.5	5
VIII. Rocky Mountains	6,262,873	111,880[e]	68	29	0	3[g]
IX. Southwest	30,432,899	819,050	51	36	4	9[g]
X. Northwest	5,634,539	220,000	89	2	9	—
U.S. Totals	180,530,874	6,856,168[g]	54	18	19	9

[a] Based on 1992 Needs Survey Data (US EPA, 1993).

[b] Includes all forms of land application practices, such as application of liquid, dewatered cake, dried, composted, alkaline stabilized, or otherwise processed product to cropland, forests, reclamation sites, lawns, parkland, etc.; use as organic fertilizer or soil amendment; use in potting mixes and the production of topsoil, etc. (including use as a daily or final landfill cover as land application).

[c] Includes co-landfilling with solid waste; monofilling; permanent disposal in piles or lagoons, etc.

[d] Shipped out of state for use/disposal (mainly by land application and landfill).

[e] Includes estimates based upon population served by POTWs and regional conditions.

[f] Some estimates (based upon the number of households served by POTWs) of total production for Pennsylvania would increase this number by 1,578,639 dmt/yr.

[g] Long-term storage.

(*Source*: US EPA.)

technologies such as these have allowed for lower water content, less odor, and easier handling of biosolids.[2]

The trends in wastewater flow increases during 1986–1996 shows an average of a 4% annual decrease in wastewater flow at POTW using only primary treatment, while wastewater flow at POTW using secondary or higher levels of treatment has increased about 2% per year. Assuming that these trends will continue into the future, and using the same methodology used to estimate 1998 biosolids generation, the expected biosolids generation amounts in 2000, 2005, and 2010 are shown in Table 65. Future biosolids production is expected to increase from 6.9 million dry tons in 1998 to 7.1 million dry tons in 2000, 7.6 million dry tons in 2005, and 8.2 million dry tons in 2010. This represents a 19% increase from 1998 to 2010.[2] These increases are largely due to anticipated increases in population served and, to a lesser extent, the increase in POTW using secondary treatment and the subsequent slight increases in quantities of biosolids produced.

Table 65. Projections of Biosolids Generation for Use or Disposal in 2000, 2005, and 2010.[2]

Year	Total (million Year U.S. dry tons)
1998	6.9
2000	7.1
2005	7.6
2010	8.2

5.3. *Use and Disposal*

Just as the generation of biosolids has increased over the past 20 years, so has the use of biosolids. There are three major reasons for this increase. First, regulatory influences on both the federal and state levels have encouraged the beneficial use of biosolids, either directly through guidance and federal policies on beneficial use or indirectly because of stringent and higher-cost requirements for disposal practices. Second, better biosolids research and technology also have helped alleviate public concern regarding the human health and environmental impacts of biosolids. Third, outreach, education, and marketing efforts have been improving public perceptions in some areas of the United States about the beneficial use of biosolids, although public acceptance problems persist in other areas.[104] The potential for a growing positive acceptance of beneficial use of biosolids could lead to increasing biosolids recovery in the future.

Disposal of biosolids is expected to decrease because of regulatory influences, voluntary improvements in biosolids quality, and the resulting increase in biosolids use. Regulatory influences include the increased restrictions on incineration, surface disposal, and landfilling in the Part 503 Biosolids Rule, the Part 258 Landfill Rule, and various state requirements, which also have driven up the costs of these disposal methods. In some municipalities, however, decreases in landfill costs are causing shifts toward increased landfilling and reductions in biosolids recycling. This trend is evident primarily among municipalities using landfills with excess capacity. The long-term effects of this factor on biosolids beneficial use are not yet known at this early stage.

As incineration becomes more costly, the disposal of biosolids through this method is expected to decrease. Incineration is a costly means of disposal and is primarily used in large urban areas. Public concern about the environmental and health impacts of incineration has made this disposal option even more costly and difficult to undertake. Public resistance to incineration is so great that no new incinerators have been built in recent years, and expansions or upgrades to existing incinerators are difficult to get approved.

Any increased costs of biosolids disposal also are expected to promote beneficial use. MSW landfills built to meet the Part 258 Landfill Rule requirements incorporate liners, gas control, leachate control, and plans and funding for monitoring and long-term care after closure, which makes them more expensive than landfills built prior to issuance of the Part 258 Landfill Rule.

5.4. *Regulations of Biosolids Disposal*

Federal regulations have been developed by US EPA for biosolids and solid waste disposal, which describes comprehensive criteria for the management of biosolids. In February 1993, the United States federal standards for the use or disposal of biosolids (Title 40 of the Code of Federal Regulations [CFR], Part 503) were enacted.[98] US EPA developed a new regulation to protect public health and the environment from any reasonably anticipated adverse effects of certain pollutants that might be present in biosolids as required by the Clean Water Act Amendments of 1987. This Part 503 rule established the requirements for the final use or disposal of sewage sludge (biosolids) when biosolids are applied to land to condition the soil or fertilize crops or other vegetation grown in the soil, placed on a surface disposal site for final disposal (landfill), or fired in a biosolids incinerator.[97] This regulation is usually referred to as Part 503 or the Part 503 rule (4).

The Part 503 rule establishes requirements for the final use or disposal of biosolids when biosolids are:

- Applied to land to condition the soil or fertilize crops or other vegetation grown in the soil.
- Placed on a surface disposal site for final disposal.
- Fired in a biosolids incinerator.

The rule also indicates that if biosolids are placed in an MSW landfill, the biosolids must meet the provisions of 40 CFR Part 258.

The Part 503 rule includes five subparts as listed below (8):

1. General provisions,
2. Requirements for land application,
3. Requirements for surface disposal,
4. Requirements for pathogen and vector attraction reduction, and
5. Requirements for incineration.

As shown in Fig. 29, for each of the regulated use or disposal practices, a Part 503 standard includes general requirements, pollutant limits, management practices, operational standards, and requirements for the frequency of monitoring, record keeping, and reporting. For the most part, the requirements of the Part 503 rule are

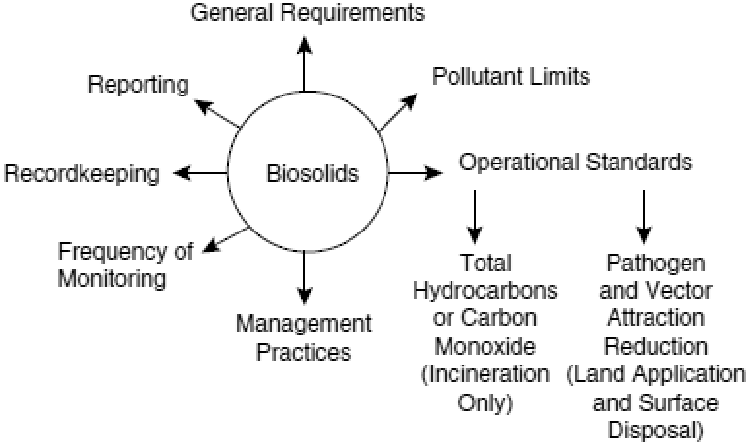


Figure 29. Part 503 Standards for Regulated Use and Disposal Practices.[98]

self-implementing and must be followed even without the issuance of a permit.[33] Subpart A of the rule covers general provisions, such as the purpose and applicability of the rule, the compliance period, and exclusions from the rule. These general provisions apply to each of the three biosolids use or disposal practices. Subpart B of the rule specifies requirements for biosolids applied to land. Subpart C of the rule covers requirements for biosolids including domestic septage placed on a surface disposal site. Subpart D of the Part 503 rule covers requirements for the control of disease-causing organisms, called pathogens, in biosolids and the reduction of the attractiveness of biosolids to vectors, such as flies, mosquitoes, and other potential disease-carrying organisms. Subpart E of the rule covers the requirements for biosolids fired in a biosolids incinerator.[33]

5.4.1. *Requirement for Biosolids Placed on a Surface Disposal Site*

Placement refers to the act of putting biosolids on a parcel of land at high rates for final disposal rather than using the organic content in the biosolids to condition the soil or using the nutrients in the biosolids to fertilize crops. Placing biosolids in a monofill, in a surface impoundment, on a waste pile, or on a dedicated site is considered surface disposal. Certain materials derived from biosolids, the quality of which has been changed by treating the biosolids or by mixing them with other materials (e.g., wood chips), are subject to the surface disposal requirements in Part 503 with one exception. If biosolids are mixed with nonhazardous solid wastes, the mixture and the land onto which the mixture is placed are subject to the solid waste regulations (40 CFR Part 258) instead of Part 503.

Table 66. EPA Pollutant Limits for Surface Disposal of Biosolids.[98]

Distance from the Boundary of Active Biosolids Unit to Surface Disposal Site Property Line (m)	Pollutant Concentration*		
	Arsenic (mg/kg)	Chromium (mg/kg)	Nickel (mg/kg)
0 to less than 25	30	200	210
25 to less than 50	34	220	240
50 to less than 75	39	260	270
75 to less than 100	46	300	320
100 to less than 125	53	360	390
125 to less than 150	62	450	420
Equal to or greater than 150	73	600	420

*Dry-weight basis (basically, 100% solid content).

5.4.2. *Pollutant Limits of Heavy Metals for Surface Disposal*

The EPA requires that one of two options for meeting pollutant limits be met for surface disposal. The first option requires that all surface disposed biosolids must not exceed maximum concentration limits of arsenic, chromium, and nickel. Table 66 shows the detail of pollutant limits.

5.4.3. *Pathogen Class and Vector Attraction Reduction for Surface Disposal*

Surface disposal of biosolids require a Class A or Class B pathogen level as defined in Section 4. The EPA requires that one of the three alternatives must be met to address pathogen reduction for surface disposal of biosolids and are listed in Table 67.

5.4.4. *Surface Disposal Alternatives*

The process of scoping potential surface disposal sites is a function of not just the quality of the biosolids to be disposed of, but also the characteristics of the proposed site. Detailed site restrictions are listed in 40 CFR Part 503, and include protection of groundwater, protection from base flood flows, regards to seismically actives site and unstable geology, protection of wetlands, collection of surface and subsurface runoff, and restriction to crop production and grazing. It should be noted that additional monitoring and reporting regulations are required, and should be referred to in 40 CFR Part 503. State regulations also apply to biosolids reuse and disposal, and can deviate from Part 503 rules from one state to another.[97]

Table 67. Pathogen and Vector Attraction Requirements for Surface Disposal.[98]

Type of Requirement	One of the Following Options Must be Met for Each Requirement
Pathogen reduction requirements	Place a daily cover on the active biosolids unit. Meet one of six Class A pathogen reduction requirements. Meet one of three Class B pathogen reduction requirements, except site restriction.
Vector attraction reduction requirements	Place a daily cover on the active biosolids unit. Reduce volatile solid content by a minimum of 38% or less under specific laboratory test conditions with anaerobically and aerobically digested biosolids. Meet a specific oxygen uptake rate (SOUR). Treat the biosolids in an aerobic process for a specified number of days at a specified temperature. Raise the pH of the biosolids with an alkaline material to a specified level for a specified time. Meet a minimum percentage solid content. Inject or incorporate the biosolids into soil.

5.4.5. *Reporting Requirements and Enforcement of Part 503*

Regulatory authorities have the right to inspect operations involved in the use or disposal of biosolids, review and evaluate required reports and records, sample biosolids at regulated facilities, and respond to complaints from persons affected by an alleged improper use or disposal of biosolids to ensure compliance with Part 503. The US EPA can initiate enforcement actions, if records are not kept or other Part 503 requirements are not met. Violations of the Part 503 requirements are subjected to the same sanctions as wastewater effluent discharge violations. The US EPA can sue in civil court and seek remediation and penalties, and it can prosecute willful or negligent violations as criminal acts. If a problem occurred, the government could seek to have the offending party correct the situation. If the US EPA is unable to take an enforcement action, Section 505 of the CWA authorizes any citizen to bring a civil action against the violator for corrective action and the same penalties that the US EPA could have sought.[97] The Part 503 rule includes reporting requirements for the following types of facilities:

- POTWs with a design flow rate equal to or greater than 1 MGD.
- POTWs that serve a population of 10,000 or greater.
- Class 1 biosolids management facilities that are POTWs required to have an approved pretreatment program (5 MGD or greater as per 40 CFR Part 403.3a) and POTWs located in states that have elected to assume local program responsibilities for pretreatment (140 CFR 403.10e), and treatment works processing domestic sewage (TWTDS) that the US EPA or the state have classified as Class 1 because of the potential to negatively affect public health and the environment.

5.5. *Biosolids Disposal Method*

Once biosolids have been processed in a waste treatment plant, a viable disposal option must be found.[5] As environmental quality standards and biosolids production increase simultaneously, municipal treatment plants are having a more difficult time finding the proper sites for disposal. With the virtual elimination of ocean dumping, there are several options available today for the disposal or utilization of biosolids:

- direct land application,
- composting and land application,
- heat drying and land application,
- incineration,
- landfilling, and
- dedicated land disposal (DLD).

5.5.1. *Direct Land Application*

Direct land application can be beneficially used in agriculture, forestry, and land reclamation. The biosolids can be applied either in a liquid form with low solids or as a semisolid following dewatering. Because of the nature of the material, it contains a large amount of water and it is often applied within relatively short distances. Direct land application for beneficial use involves some form of partial stabilization such as digestion or alkaline stabilization. Digestion results in US EPA 503 Class B biosolid whereas alkaline stabilization can result in a Class A or B product. The US EPA 503 regulations allow for either a Class A or Class B to be applied to land.[97]

5.5.2. *Composting and Land Application*

Composting is the biological decomposition of sludge under conditions that allow the development of thermophilic temperatures resulting from biologically produced heat. Composting appears to be a viable alternative for sludge treatment since it reduces the load of pathogenic microorganisms, yielding a final product rich in organic matter and nutrients that can be used as a soil supplement for different horticulture, agricultural, or land application purposes. Besides, it contributes to the suppression of certain plant diseases caused by several pathogens. However, this method displays several restrictions related to heavy metal concentration and moisture present in the sludge.

5.5.3. *Heat Drying and Land Application*

Heat drying is a form of land application. This technology produces a US EPA 503 Class A product that provides for a wider range of uses. Today, most of these products meet the EQ criteria for trace elements. The drier nature of these two materials provides for bagging and shipping to greater distances. Heat-dried biosolids are considered as a fertilizer and applied at fertilizer rates. They may be used as a supplement to other inorganic fertilizer material to increase the plant nutrient content.

5.5.4. *Incineration*

Biosolids incineration requires extremely high temperatures (450°C and higher), which are provided by specialized furnaces. The two most common types are multiple-hearth furnaces and fluidized-bed furnaces. However, the latter is preferred because in general it produces fewer emissions. In order to prevent dangerous gases, particles, and other pollutants from escaping into the environment, these furnaces are equipped with air pollution devices. The most common emission controllers include wet scrubbers, fabric filters, afterburners, and precipitators. After incineration, only about 20% of the biosolid volume remains, and many of the volatile organic chemicals and pathogens are eradicated. Advantages and disadvantages of incineration are listed in Table 68.

5.5.5. *Landfilling*

About 17% of the total application of biosolids today is from landfill and surface disposal. The use of landfills reduces treatment costs for many municipal wastewater treatment facilities because landfilled biosolids require less advanced treatments. Federal and state regulations do not require many of the stabilization and dewatering methods for landfilled biosolids.

Landfilled biosolids also require no additional monitoring for leachates and metal content that land disposal option for increasing amounts of biosolids. However, bans on disposal of biosolids in landfills, landfill capacity concerns, and landfill closures have greatly hindered the use of landfills as a profitable disposal option. Limited space means higher tipping fees and a greater cost to the wastewater facility trying

Table 68. Advantages and Disadvantages of Incineration.[2]

Advantages	Disadvantage
Reduces volume of biosolids allowing easier disposal.	Dewatering is required before beginning the incineration process because water content increases the energy necessary.
Energy can be collected from the incineration process.	Pollution devices must be installed to capture emissions.
Destroys virtually all of the pathogens and VOC present.	The ash produced has large heavy metal concentrations.
Requires little land area for operation relative to landfilling and dewatering.	High energy requirements and costs.

Table 69. Advantages and Disadvantages of Landfills.[2]

Advantages	Disadvantage
Once disposed of in a landfill, there are no additional costs for monitoring and no concern over human contact.	Does not utilize the nitrogen and nutrient rich qualities of the biosolids.
Leachates are contained safely inside the landfill's liner.	Landfill space has become extremely limited in recent years and biosolids take up space when they could be used beneficially in the environment.
Public concern and outcry is avoided.	Involves a large initial contracting cost for transportation and tipping fees.

to dispose of its biosolids inventory. Table 69 summarizes the pros and contras of disposing of biosolids in landfills.

Surface disposal is another option to be considered by a municipal wastewater facility. The Biosolids Rule (40 CFR 503) defines surface disposal as the placement of biosolids for final disposal on land on which only biosolids are present. Under this rule, the biosolids must remain untouched on the land for at least two years to be considered disposed; otherwise, it is classified as storage or treatment. Site selection for biosolid surface disposal is based on factors such as land slope, soil conditions, and required minimum proximity to ground or surface water. Human and animal contacts also weigh in heavily when trying to select an appropriate site. The main difference between surface and land disposal is application rate. Surface disposal simply means that the biosolids have been applied at levels greater than agronomic rates and require a form of leachate monitoring such as underground wells.

The most common form of surface disposal is depositing biosolids in monofills. Monofills come in various forms including excavated trenches, large excavated areas, mounds, and layers on the ground covered with topsoil. Upon completion, all monofills are covered with several centimeters of soil to prevent odor and reduce human and animal contact.

5.5.6. *Biosolids Disposal at Dedicated Facilities*

A number of agencies have elected to disposal of biosolids at "dedicated" facilities. These include:

- mono fills,
- land disposal sites, and
- disposal lagoons.

Biosolids-only landfills are defined as disposal sites that are used exclusively for the disposal of biosolids. The application of the biosolids can be in either the dewatered or liquid state. However, the biosolids are usually applied in the dewatered state. DLD sites are those sites where the biosolids are applied to the surface of the land on a routine basis where the objective is disposal rather than use. At DLD sites no crops are grown. Sites normally employ annual application rates of 50–100 dry tons per acre annually. The biosolids are usually applied as a liquid.

Containment of biosolids landfills is of extreme importance. The most cost-effective landfills are those where the natural soils at the site support such a development. Lagoon disposal is a method available that stabilizes biosolids through anaerobic endogenous respiration while controlling odors using an aerobic water seal. Biosolids are typically introduced to the bottom of a 15–20-foot-deep lagoon. The biosolids and anaerobic liquid are capped by an aerobic zone near the water surface. The aerobic zone is maintained by natural convection and surface aeration, a facility operating a disposal lagoon in central California reports a 45% reduction of volatile solids after one year of storage.

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Chapter 2

SLUDGE MANAGEMENT

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Abstract

Sludge accumulates as a residue in all wastewater treatments. Sludge contains the solids and colloids separated from wastewater, as well as substances from biological and chemical operation units. Sludge contains all of the impurities gathered from water or wastewater treatment processes, and thus has a variable nature and must be treated accordingly. This chapter summarizes all aspects of sludge management in practice.

Keywords: Sludge dewatering, stabilization, disposal, landfill, reuse, incineration, management.

1. The Origin of Sludge

Sludge accumulates as a residue in all wastewater treatments. Sludge contains the solids and colloids separated from wastewater, as well as substances from biological and chemical operation units. The precise definition of "sludge" is vague. For example, Vesilind defined sludge as "the residual from treatments of wastewater that is semisolid, odiferous, unmanageable, and dangerous."[1] Moreover, in 1991, Metcalf and Eddy defined sludge as the residual "resulting from wastewater treatment processes, generally taking the form of liquid or semisolid liquid that typically contains 0.25–12% solid."[2] In 2000, a report from International Water Association defined sludge as "a mixture of water and solids separated from various types of water through natural or artificial processes." In fact, sludge contains all of the impurities gathered from water or wastewater treatment processes, and thus has a variable nature and must be treated accordingly.

Some sludge is produced during wastewater treatment, including the primary sludge that comprises settleable solids removed from the primary clarifier and the waste biological sludge that comprises biological solids generated in secondary wastewater treatment plants (Fig. 1). In 2002, annual sewage sludge production rate was approximately 8,000,000 dried tons both in the United States and in the European Union, 2,100,000 dried tons in Japan, and 1,000,000 dried tones in Canada. Moreover, the rapidly developing sewage treatment industry in China is expected to see annual sewage sludge production.

In the drinking water industries, the coagulated solids-coagulant matrix is separated from the water stream, collected and termed the "water treatment residue," or the "water sludge." Additionally, numerous kinds of industrial sludges are generated from treating different industrial wastewaters. The industrial sludges are frequently classified as "hazardous waste" requiring special and intensive treatment.

Table 1 summarizes the characteristics of the typical sludge samples. The implementation of treatment processes to determine the specific purpose of sludge disposal or reuse options is termed "sludge management." This chapter introduces the processes and method of synthesis of sludge management. Useful information could be found in Refs. [3–11].

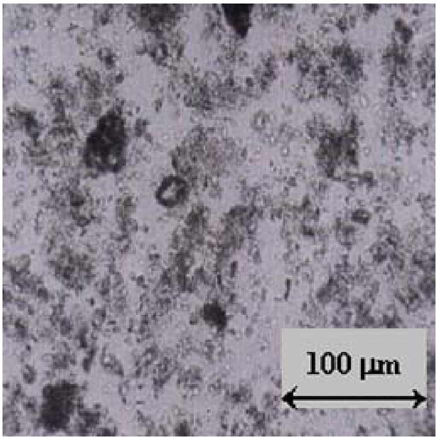
2. Sludge Disposal

Conventional sludge disposal methods are all problematic, as follows:

- Land/sea disposal (outlawed during the past decade).
- Sanitary landfill (new sites are difficult to establish and the practice is controversial).

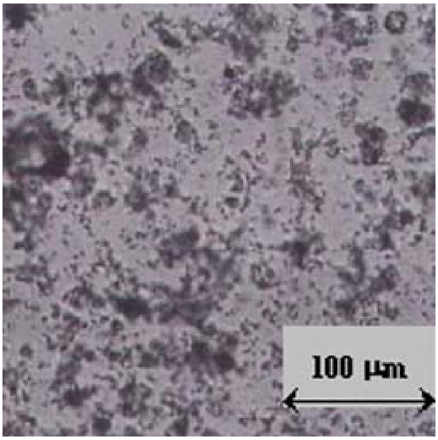
Alum sludge

- Collected at PingTsan Water Works, Taiwan, using PACl as coagulant
- Dried solid content 20,000 mg/L
- pH: 6.0–6.5
- Floc size: 20–30 μm
- Zeta potential: -10 to -5 mV
- SCOD: <10 mg/L
- ECPs: <0.5 mg/g DS
- SVI: 20–40



Pulp and paper sludge

- Collected at Young-Feng-Yu Pulp and Paper Plant
- Dried solid content: 6,800–7,200 mg/L
- pH: 6.3–6.7
- Floc size: 20–30 μm
- Zeta potential: -18 to -15 mV
- SCOD: 50–80 mg/L
- ECPs: <1 mg/L
- SVI: 40–60



Copper sludge

- Collected at the Electroplating Plant
- Dried solid content: 10,000–11,000 mg/L
- pH: 5.1–5.6
- Floc size: 100–120 μm
- SCOD: <1 mg/L
- ECPs: ND
- SVI: 50–80

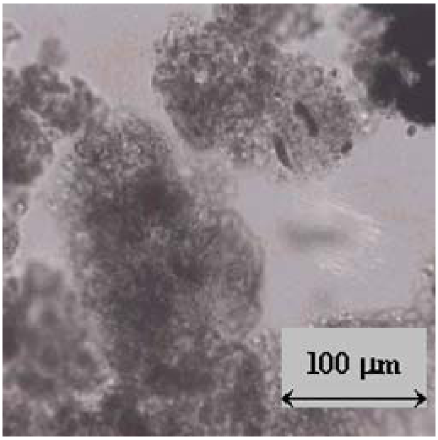
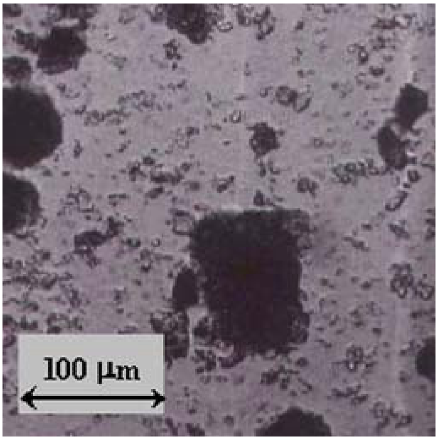


Figure 1. The Appearances of Typical Sludge Samples.

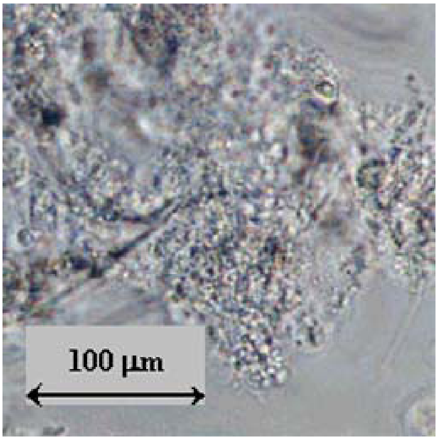
Scum sludge

- Collected at flotation unit of a fiber plant
- Dried solid content 17,000–18,000 mg/L
- pH: 5.8–6.1
- Floc size: 25–35 μm
- Zeta potential: -17 to -14 mV
- SCOD: 950–1,000 mg/L
- ECPs: ND
- SVI: 80–120



Activated sludge

- Collected at food-manufacturing plant
- Dried solid content: 7,500–8,500 mg/L
- pH: 6.7–6.9
- Floc size: 80–100 μm
- Zeta potential: -18 to -13 mV
- SCOD: 20–50 mg/L
- ECPs: 10–12 mg/g DS
- SVI: 80–100



Activated sludge

- Collected at a petrochemical plant of Taiwan
- Dried solid content: 10,000–14,000 mg/L
- pH: 7.1–7.3
- Floc size: 40–50 μm
- Zeta potential: -20 to -16 mV
- SCOD: 70–100 mg/L
- ECPs: 15–20 mg/g DS
- SVI: 90–120

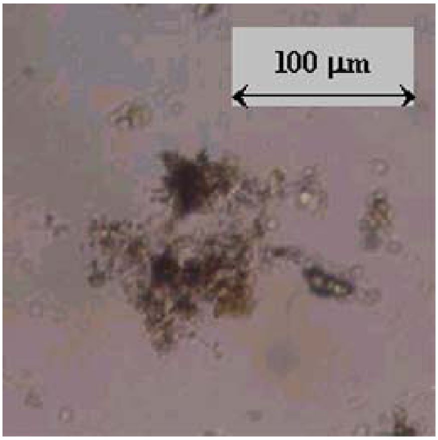


Figure 1. (*Continued*)

Table 1. Characteristics of Typical Sludges.

Sludge	Solid Fraction (%)	Characteristics
Primary (raw)	4–8	Gray-brown, odorous, vile, and mechanically dewaterable
Activated sludge	1–2	Little odor, yellow-brown, and high biological activity
Anaerobically digested	6–8	Drain well on drying bed, musty, black, and little odor
Aerobically digested	2–4	Yellow-brown
Alum sludge[a]	1–2	Gray-yellow, odorless, and difficult to dewater
Copper sludge	3–5	Black, alkali suspension, and hard to dewater

[a] Becomes very objectionable for eutroficated water source.

- Composting and land application (high failure rate for full-scale application, odor problems, and lack of a compost market).
- Incineration (high energy costs and need for ash treatment and flue gas treatment).

Increasingly stringent regulations and strong public objections increase the difficulty of conventional disposal routes. Management costs associated with such routes increase significantly with time. Traditionally, sludge was disposed of from wastewater treatment plant without special care. Nowadays, sludge disposal is done more carefully and sludge treatment/disposal accounts for 50–60% of the capital and operational costs of full-scale wastewater treatment plant.

Dumping of sewage sludge in the ocean is now almost forbidden. The primary methods of sludge disposal include landfill, land application, and incineration. US Regulations regarding the land application of sewage sludge are summarized in 40 CFR Part 503. Meanwhile, EU regulations on sludge disposal include the Sewage Sludge Directive 86/278/EEC, the Organic Farming Regulation (EEC) n. 2092/91, the Landfill Directive 1999/31/EC, and the Commission Decision 2001/688/EC concerning the eco-labeling of soil improvers and growth media. EU regulations on sludge disposal are currently being revised and tightened.

A recent review demonstrated that most developed countries used the following three disposal routes: sludge landfill, sludge incineration (followed by ash treatment and disposal), and land use. Table 2 compares the popularity of these three sludge disposal routes in the 12 original member countries of the European Union. Landfill apparently has become less popular while sludge incineration has increased significantly. However, five EU member countries have never considered incineration as a sludge disposal route. Table 3 lists the disposal routes used in certain countries. Belgium incinerated more sludge in 1998 than in 1990, by reducing the use of land

Table 2. Final Disposal Routes of Sewage Sludge by the 12 Old Member Countries of European Union.

	1984	1992	1999	2005 (est.)
Reuse (%)	37	39	40	45
Incineration[a] (%)	9	11	24	38
Landfill (%)	54	50	36	17

[a] Five member countries, Greece, Finland, Ireland, Luxemburg, and Sweden completely excluded incineration as one of the final disposal options.

Table 3. Sludge Disposal Options for Some Countries.

Country	Landfill (%)	Incineration (%)	Agriculture (%)	Reclamation (%)	Others (%)
Belgium[a]	43	—	57	—	—
Belgium[b]	58	23	13	—	—
Denmark[a]	29	28	43	—	—
Denmark[b]	12	21	67	—	—
Japan[c]	21	61	12	—	7
The Netherlands[a]	63	6	30	—	—
South Africa[a]	67	—	28	—	—
Singapore[a]	—	—	—	100	—

[a] year 1990; [b] year 1998; [c] year 1995.

application. In Denmark, on the other hand, land application of sludge was more popular. Consequently, the disposal route for sludge from different countries/regimes has not "converged" on a single and universal solution. Instead, the routes adopted by nearby countries/regions depend heavily on local circumstances and change over time. Restated, no universal solution to the sludge disposal problem exists, and instead "localized" optimal solutions should be sought.

A typical sludge treatment/disposal system comprises the following four stages: (1) pretreatment stage, during which sludge characteristics are altered to enhance subsequent process performance; (2) dewatering stage, during which moisture is separated from the sludge body to reduce sludge volume, (3) post-treatment stage, for sludge stabilization or detoxification, and (4) disposal stage, for achieving safe and economical sludge disposal. Sludge management system optimization aims to deal with sludge to maximize the recycle/recovery benefits, be appropriate to local circumstances, link wastewater services to other waste management services via integrated planning, and assure long-term service. Until now, global process optimization for sludge management has been based on heuristics and experience.

3. Making Management Systems

3.1. *System Selection*

Sludge production increased when new wastewater treatment facilities were installed or existing facilities upgraded. Meanwhile, the newly enforced acts frequently made existing treatment/disposal options impractical. Therefore, engineers often must decide what should be done for their sludge to meet the updated regulations cost-effectively.

The selection of an appropriate sludge management system depends on several factors, mostly determined by technical and economical circumstances. Poor selections are frequent, mostly because of the lack of a systematic approach to a rational process. Figure 2 displays the workable decision-making route for a rational management process. To establish a sludge management plan, information first is required on the quantity and quality of sludge produced during the expected service period. The final disposal route then is selected based on the quality and quantity of the sludge produced. Finally, the appropriate treatment chain is selected to bring the raw sludge to the final disposal site in the safest, most economical, and most publicly acceptable way.

Figure 3 illustrates a typical sludge treatment process. Taipei City has been dealing with its sludge: thickening → dewatering → landfill. However, this is definitely not a sustainable management option since Taiwan lacks landfill sites. In comparison, Singapore uses thickening → digestion → dewatering → land reclamation. Meanwhile, Tokyo treats sludge using thickening → digestion → dewatering → drying → incineration → reuse of ash to produce construction materials. Options from one country are not necessarily suitable for other countries/regimes. For instance, current regulations in Taiwan prohibit sludge incineration.

The following sections discuss in detail the three stages of the process for managing sludge synthesis, displayed in Fig. 2.

3.2. *Sludge Characteristics*

Sludge treatment/disposal involves separating water from the sludge body, treating the separated water as normal wastewater, drying the solid residual as much as

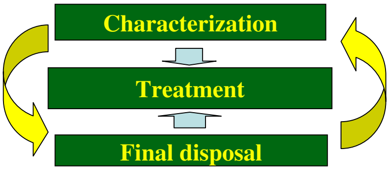


Figure 2. The Decision-Making Route.

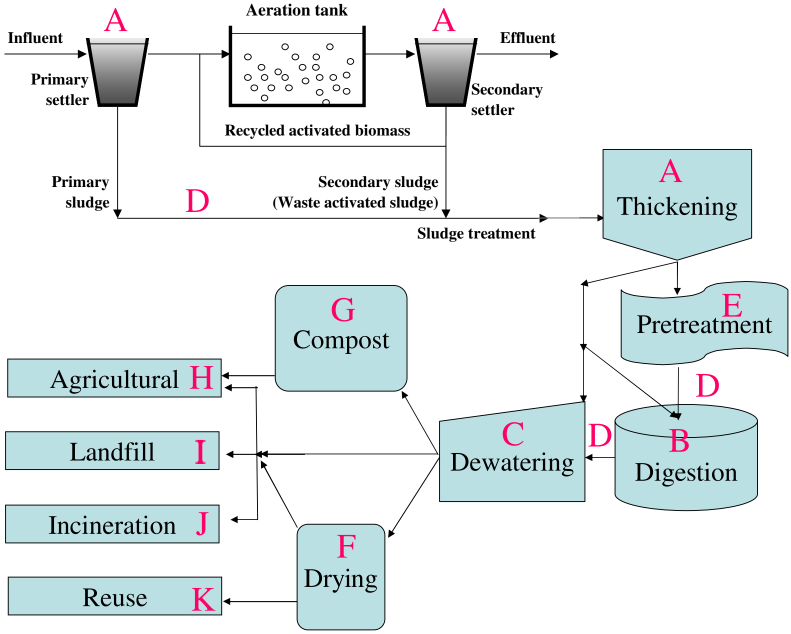


Figure 3. A Sludge Management Network and the Required Physical Characteristics.

Table 4. Required Property for Sludge Treatment Process.

Property	Process	Property	Process	Property	Process
Solids fraction	A–K	Rheology	D	Heat value	J
Volatile fraction	A–K	Filterability[b]	C	Fertilizer value	H
Floc size	A, C	Shear index	I	Heavy metal	H, K
Floc strength	C	Zeta potential	A, C	Pathogen	H
Settleability[a]	A	pH, alkalinity	B	Stability	H, I

[a] Characterized by zone settling velocity (ZSV) or sludge volume index (SVI).
[b] Characterized by capillary suction time (CST), time to filter (TTF), or specific resistance to filtration.

possible, and finally treating it as municipal solid waste (MSW). Therefore, the sludge manager should characterize the water-side, solid-side, and their interfaces. On the water-side, commonly measured quantities include the pH, temperature, SS, BOD, COD, TOC, and so on. On the solid side, the solid content, heat value, heavy metal content, chemical compositions, pathogen levels, and so on should be measured. Finally, on the interface, the interfacial area, surface charge, hydrophobicity, and so on should be characterized.

Table 4 lists the commonly accepted "important quantities" that control the specific sludge management process. The capital letters in Fig. 3 denote the

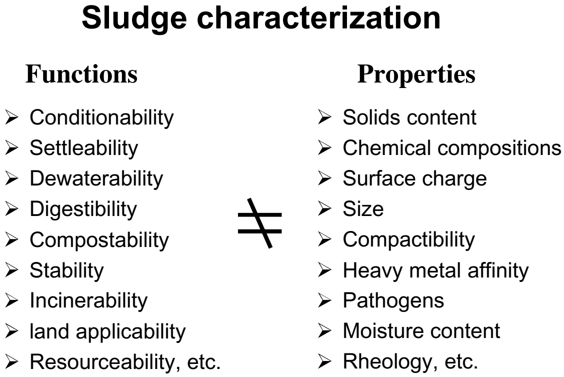


Figure 4. Sludge Properties Do Not Necessarily Correspond to Its Functions.

corresponding processes listed in Table 4. For example, the solids fraction is required for forecasting the performances of all of the mentioned processes. Meanwhile, pathogen level is significant for agricultural uses, but is insignificant for sludge-drying efficiency.

A major difficulty in sludge characterization is that the measured properties do not directly correlate with the process performances, or the process "functions." For instance, even if the sludge properties are known (Fig. 4), including solid content, chemical compositions, zeta potential of flocs, and so on, forecasting sludge dewatering speed in a centrifuge remains impossible. Furthermore, normally measured properties are insufficient for estimating whether a specific wastewater sludge can be converted to compost using a static pile. Until now, engineers have relied on heuristics to bridge this gap, mostly based on previous experience accumulated by design firms. Designing a process without conducting sufficient tests on the real sludge is very risky. For example, eight years ago when planning to increase sewage system coverage, Taipei City recognized the need to treat increased sewage sludge production. The city thus designed and built egg-shaped anaerobic digesters to treat the sewage sludge, designing these facilities based on the properties of "nonexisting" sludge. However, the digesters have remained inactive since their completion in 1998, because of the feed sludge characteristics differing from those used for the original design. This kind of failure causes major investment loss, owing to insufficient sludge characterization. Engineers thus frequently must conduct at least pilot scale tests to confirm the process functions.

Table 5 lists certain process functions that require characterization before implementation in the field. The commonly measured properties and the frequently adopted indices are also listed in the table.

Table 5. Typical Functions of Sludge.

Function	Factors to be Considered	Index/Implication
Flowability	Dried solids content, zeta potential, floc size distribution, surface hydrophobicity, viscosity, and dissolved organics	1. Choice of coagulant 2. Pumping characteristics 3. Link to sludge dewaterability
Settleability	Dried solids content, used coagulant, zeta potential, floc size, viscosity, floc shape, content of filamentous bacteria, and sediment height	1. Supernatant turbidity 2. Sediment compactibity 3. Sediment concentration
Dewaterability	Dried solids content, dewatering device, applied pressure difference, compressibility, viscosity, ease of filter clogging, and adequate preconditioning stage	1. Filtration resistance 2. Residual moisture content 3. Necessity of subsequent drying stage
Digestability	Dried solids content, floc size, heavy metal content, organic matter content, sludge age, and aeration amount (aerobic)	1. Biogas production rate 2. Pathogen content 3. Liquor COD 4. Dewaterability
Landfillability	Dried solids content, slope stability, organic content, land price, and regulations	1. Biogas production rate 2. Leachate quality 3. Groundwater quality 4. Vector attraction
Land applicability	Dried solid content, fertilizer value, pathogen level, heavy metal content, odor potential, methane production potential, and groundwater table	1. Market of sludge fertilizer 2. Soil quality 3. Plant growth
Incinerability	Water content, volatile solids content, heat value, chlorine content, and fuel price	1. Flue gas emission control 2. Disposal route of ash 3. 產生之廢水性質與流选.

4. Sludge Disposal Routes

Sludge consists of a mixture of useful materials and pollutants. A current trend exists to seek ways to recycle useful materials from waste, before disposal through landfill, incineration, or other methods. Table 6 lists the typical chemical elements of sludge. Biological sludge has higher levels of N and P than primary sludge, owing to the bioaccumulation effect during wastewater treatment. Digestion stabilizes sludge by removing part of biomass from the sludge body, thus reducing sludge organic fractions. However, the organic matters removed during digestion normally are highly volatile and cause odor problems for undigested sludge. Furthermore, the trace contaminants in sludge are the key determinants of sludge suitability for land application.

Table 6. Elements in Sludges.

Type of Sludge	Carbon (%)	Nitrogen (%)	Phosphate (%)
Primary sludge	30–40	4–6	0.5–3
Biological sludge	35–45	8–12	2–6
Digested sludge	20–30	4–5	2–3.5

As mentioned above, the primary sludge disposal routes identified by a survey of 24 countries worldwide include landfill, land application, and incineration. Meanwhile, another option is to reuse the sludge as a raw material in other industries, such as the construction industry, or the cement industry, although the market for such recycled sludge remains limited to date. Among the three main disposal routes, the use of sludge on land is the most tightly regulated, owing to its potential for long-term environmental damage. Generally, sludge must be stabilized before land application. Recent EU regulations have required the pre-stabilization of sludge disposed of through land filling.

The following sections briefly summarize the characteristics of the major final disposal routes for sludge.

4.1. *Landfill*

Sanitary landfill is the oldest and most popular disposal route. The common practice is to co-landfill the sludge together with the MSW at the landfill site. This option recently has lost favor because of the negative image of sludge as a pollutant: leaching to pollute surface and groundwater and emitting biogases to contribute to the greenhouse effect. Consequently, the practice of sanitary landfill is under pressure to change, both quantitatively and practically.

In the United States 1,200,000 dried tons of sewage sludge were sent to landfill in 1998, with another 200,000 tons of sludge being used to the landfill cover. Since regulations require close monitoring of groundwater for pollution prevention, landfill has become more expensive in the United States than land application when the sludge involved is clean. Sludge landfill is regulated by 40 CFR Part 503 of USEPA "Standards for the use or disposal of sewage sludge."

In Europe, six countries contribute over 50% of total sludge landfill disposal in the European Union. The practice is regulated by Council Directive 1999/31/EC on the Landfill of Waste. However, many individual member countries implement stricter management strategies. Notably, both Greece and France banned sludge landfill in 2002. Moreover, Denmark and Germany established limits on disposal of organic materials in landfill sites. Additionally, Italy allows co-landfill of dewatered and

Table 7. Advantages and Disadvantages of Sludge Landfill.

Advantages	Disadvantages
• Co-disposal with MSW: good practice • Make reclaimed land	• Landfill is being phased out, or increasingly more expensive • Leachate has to be properly treated • Public concern and objection: political problem • Groundwater monitoring needed • Greenhouse gases and global warming effects • Odor and vector control

stabilized sludge with MSW only after dewatering and stabilized. Furthermore, the Netherlands limited the maximum water content in the landfilled sludge as of 10%.

In Asia and Australia currently landfills 76% of its sewage sludge, although Sydney landfills only 19%. Meanwhile, Yokohama and Sapporo makes no use of landfilling. However, in Taiwan, all sludge presently is landfilled.

Co-disposal of sludge with MSW or soil has high volume efficiency because the former can occupy the void space in the landfill. However, owing to the easy biodegradation of sewage sludge, co-disposal is characterized by a strong odor and leachate potential. The typical application rate for sludge landfill ranges 1,000–8,000 m^3/ha, with a minimum solid content of 20%. The key parameters include sludge rheological properties and solid content. The volatile solid contents determined the odor and methane production potentials.

Table 7 lists the advantages and disadvantages of sludge landfill processes.

4.2. *Incineration*

The first sludge incinerator was installed in Michigan in 1934, and incineration remained a widespread sludge disposal method until the 1970s. However, rising energy costs caused incineration to become the least preferred option by the 1980s. Nowadays, modern incinerators have reduced the energy consumption associated with incineration, landfill costs are increasing, and regulations governing land disposal of sludge are tightening, meaning incineration once again is becoming a major route of sludge disposal, particularly for densely populated and urban areas.

Incineration requires that sludge temperature be increased to 100°C to evaporate water from the sludge, after which the water vapor temperature, air temperature, and solid phase must be raised to ignition point. This process is energy-intensive, and consequently sludge heat value is the key consideration in sludge incineration. Sludge organic fraction has a heat value of 25 MJ/kg-volatile matters. Given 40% inert fraction in solid phase and 80% residual moisture in the dewatered cake, the heat value of wet cake is reduced to 3 MJ/kg-dewatered cake, well below that for the fossil fuel (40 MJ/kg). The moisture in the dewatered cake should be below 60% by weight to enable self-sustainable incineration. However, this value is significantly

Table 8. Advantages and Disadvantages of Sludge Incineration.

Advantages	Disadvantages
• Great volume reduction • Phosphate recycling • High contaminant destruction ratio • Minimization of waste transport and disposal • Most suitable to densely populated countries	• Causes air pollution problem and requires control of heavy metals, dioxins, etc. • Energy-intensive process • High capital cost • Testing, operating, and controlling the process are labor-intensive and expensive • Ash must be disposed of • Depleting natural resources • Greenhouse gases and global warming effects

below that achievable by most mechanical dewatering devices. Auxiliary fuel thus is frequently required for sludge incineration.

Currently, the most popular sludge incinerator type is the fluidized bed furnace (FBF), which achieves good thermal efficiency by violently mixing sand particles and the burned sludge. Additionally, the high heat capacity of the sand bed can achieve quick start-up.

Table 8 lists the advantages and disadvantages of sludge incineration processes.

4.3. *Land Applications*

Land application is the only (possibly) sustainable method of sludge disposal. The sludge could be applied to agricultural land, forestland, reclamation sites, and public contact sites. All these applications are subjected to stringent regulations, such as 40 CFR Part 503 by US EPA.

Table 6 lists the chemical elements of sludges. The sludge contains plant macronutrients, N and P, and various micronutrients. The nutrients in sludge can be used as fertilizers. The use of sludge as soil conditioner can make the latter porous for root growth. Sludge application resembles conventional faring operations. Notably, excessive sludge application pollutes the ground or surface water with NO_3^--N. The main concern in sludge land application is the emergence of new and highly toxic chemicals. The toxicity can cause loss of crop yield (phytotoxicity) or affect the health of animals consuming crops planted on the land (zootoxicity). The heavy metals in sludge tend to accumulate in the amended soil. The soluble part of pollutants can easily enter the groundwater. The pathogens in sludge represent a public health hazard if they are transferred into the food chain by land application of sludge, to surface water by runoff of land sites, or elsewhere by vectors.

Pretreatment influences the agronomic value of sludge. For example, dewatering reduces soluble nitrogen compounds and drying evaporates out ammonium vapor from sludge.

Applying sludge on poor forestland or disturbed land can provide sufficient nutrients for plant growth. Additionally, industrial plants could be grown using sludge without public health concerns, meaning that application rates can be very high.

Land application of sludge pollutes soil, including pathogens that present medium-term public health risks and heavy metals that present long-term risks. Limits on heavy metal contents for land application differ between countries (Table 9), meaning that a sludge that is considered safe for land application in one country may be classified as hazardous waste in another country. In some EU countries, such as the Netherlands and Sweden, regulations are now so stringent that land application has become impractical. Table 10 lists the criteria for land use by US EPA, by classifying the sludge into Class A or Class B sludges. Class A sludge

Table 9. Typical Heavy Metal Limits for Land Application of Sludge.

Country	Cd	Cu	Cr	Ni	Pb	Zn	Hg
Brazil	20	1000	1000	300	750	2500	16
China ($pH < 6.5$)	5	250	600	100	300	500	5
($pH > 6.5$)	20	500	1000	200	1000	1000	15
Chili (agriculture)	8	1000	NA	80	300	2000	4
(eroded soil)	40	1000		420	400	2800	20
Denmark	0.5	40	30	15	40	100	0.5
France	2	100	150	50	100	300	1
Finland	0.5	100	200	60	60	150	0.2
Italy	3	100	150	50	100	300	NA
The Netherlands	0.8	36	100	35	85	140	0.3
South Africa	15.7	50.5	1750	200	50.5	353.5	10
United Kingdom	3	135	400	75	300	200	1
The United States	85	4300	NA	420	840	7500	57

Table 10. Criteria for Sludge Land Use by US EPA.

Sludge	Class B	Class A
Criteria	• Fecal coliform $<2 \times 10^6$ MPN/g-TS	• Fecal coliform $<10^3$ MPN/g-TS • *Salmonella* spp. <0.75 MPN/g-TS • Helminth Ova <0.25 egg/g-TS • Virus <1 PFU/g-TS
Processes	• Composting >40°C for 5 days and >55°C for 4 hours • Air drying for 3 months • Aerobic digestion >40 days at 20°C or >60 days at 15°C • Anaerobic digestion >15 days at 35–55°C or 60 days at 20°C • Lime stabilization at $pH > 12$	• Composting at 55°C for 3 days in vessel of in static pile, or for 15 days in windrows • Heat drying to moisture <10% • Heat treatment >180°C for 30 min • Aerobic digestion at 55–60°C for 10 days • Irradiation >1 Mrad • Pasteurization >70°C for 30 min

Table 11. Advantages and Disadvantages of Sludge Land Application.

Advantages	Disadvantages
• Old solution and good practice experience • Lower cost compared with landfill or incineration • Could be applied on agricultural field or in forest • Liquid sludge application reduces the use of chemical fertilizer	• Lacking available land • Public concern and objection: political problem • Containmented sludge has to be cleaned up by removing pathogen and heavy metals (very costly) • Sludge transportation cost become very significant if the transportation distance >100 km

can be applied to land in the United States without restriction. Although the main trend in the European Union is to spread more "cleaned" sludge on land, Switzerland plans to end the agricultural use of sludge by 2005 regardless of its cleanliness.

Table 11 lists the merits and demerits of sludge disposal through land application.

4.4. *Sludge Reuse*

Mainstream opinion regarding sludge reuse holds that it can save resources and energy, and hence is beneficial to the environment. Dewatered sludge could be melted to produce slag or tile, pyrolyzed or dried for fuel, or applied as an alternative cover for landfill sites. Meanwhile, the ash from incineration plant could be used to create lightweight aggregates, molded into bricks, added to cement, vitrificated into marble, and so on. Figure 5 illustrates some coarse aggregates made by sintering sludge/clay mixtures. Table 12 lists some sludge reuse routes.

Although the general belief is that the recovery of materials from waste serves useful resources and benefits the environment, the truth is often that converting

Figure 5. The Coarse Aggregates Made by Sintering Sludge/Clay Mixtures at 1,000°C.

Table 12. Resource Recovery from Sludge.

Sludge State	Treatment	Product	Usage
Concentrated suspension	Digestion	Biogas	Fuel
Dewatered cake	Drying	Dried cake	Fuel, fertilizer
	Fermentation	Compost	Fertilizer, soil amendment
	Sintering/melting	Slag, aggregates, tiles, etc.	Construction materials
Ash	Sintering/melting	Slag, aggregates, tiles, etc.	Cement filler, construction materials, landfill cover, soil substitutes

sludge into useful resources generally consumes more energy/materials than that can be recovered from the waste sludge. Therefore, such application is not sustainable, and this situation will persist unless some major technological breakthrough is made.

5. Sludge Management Chain

Once the sludge has been properly characterized and the final disposal route has been selected based on local circumstances, the sludge treatment train can be synthesized according to the route-map in Fig. 2. Table 13 lists the typical current sludge treatment processes. The characteristics of these processes, including the capability to reduce sludge volume, degree of stabilization, potential to land use, associated costs, and public acceptance, are also briefly summarized. Apparently, these processes are "borrowed" from wastewater treatment processes or solid waste treatment processes. The treated sludges have very different characteristics that are suitable for various end uses.

The treatment processes include conditioning, dewatering, stabilization, thermal treatment, and others, which are briefly summarized in the following.

Colloidal particles in sludge can remain stable owing to steric hindrance or charge repulsion. Coagulation or flocculation using chemicals is frequently used to enlarge the floc size or to compress the floc interior to facilitate solid–liquid separation efficiency. The widespread used coagulants include inorganic metal salts or organic polyelectrolytes. Ferric salts are the most popular inorganic coagulant in North America. Meanwhile, in the rest of the world, aluminum salts such as alum ($Al_2(SO_4)_3 \cdot 14 - 18H_2O$) and aluminum chloride ($AlCl_3$) are the most commonly used. This adsorption and charge neutralization mechanisms facilitate particle aggregation. Precipitation occurs when the coagulant dose exceeds total metal hydroxide solubility. The precipitated floc can sweep over the suspension and trap and remove the fine particles during settling. This mechanism is termed sweeping enmeshment. Organic polyelectrolytes include the anionic polyelectrolytes, cationic

Table 13. Typical Sludge Treatment Processes.[a]

Process	Volume Reduction	Stabilization	Land Use	Capital Cost	O&M Cost	Energy Recovery	Public Acceptance
Dewatering	H	NA	NA	M	M	NA	H
Drying bed	H	NA	Class B	M	L	NA	L
Thermal drying	H	H	Class A	HH	HH	M	L
Thermal treatment	NA	H	Class A	HH	HH	M	L
Lime stabilization	Negative	H	Class B	L	M	L	H
Aerobic digestion	NA	H	Class A	M	HH	NA	L
Anaerobic digestion	NA	H	Class B	H	L	H	H
Compost	Negative	H	Class A	M	H	NA	M
Incineration	HH	HH	NA	HH	HH	H	L
Wet air oxidation	NA	H	Class A	HH	H	M	H
Thermal pyrolysis	H	H	NA	HH	H	M	H
Melting	H	H	NA	HH	HH	M	H
Freeze/thaw	H	H	Class B	L	L	NA	M

[a] HH: very high; H: high; M: medium; L: low; NA: not applicable; Negative: negative effect.

polyelectrolytes, and nonionic polymers. Two mechanisms, charge neutralization and bridging, mainly correspond to the observed flocculation behavior between the polyelectrolyte and the suspended particles. Owing to the high cost of flocculant, dual conditioning, namely, one flocculant combined with other conditioner (inorganic flocculant or other flocculant with different molecular weight or charge density) could help improve the dewaterability. Addition sequence may influence floc formation.

Sludge dewatering reduces the sludge volume to facilitate the subsequent treatment/disposal processes. Sludge thickening/dewatering processes could be classified into "mechanical" and "nonmechanical" processes. More than 75–80% of moisture removal occurs during the thickening stage. Sludge thickeners include the gravity thickener, centrifugal thickener, and flotation thickener. The drying bed dewaters sludge through gravity draining and air evaporation. The sludge can be dewatered to 60–70% solids in 6–12 months. The belt filter press contains an endless filter belt and a press belt, which have numerous "bends" for saving the footprint of the device. The sludge could be dewatered to 20% solids for activated sludge and 25–30% for raw primary sludge. The plate and frame press consists of vertical plates covered with a filtering medium held side by side in a frame. The sludge can be dewatered to 15–20% solids for activated sludge. The centrifuge uses centrifugal force to increase the sedimentation rate and enhance dewatering of sludge. High-solid centrifuge can reach a high solid content in dewatered cake.

The commonly adopted sludge stabilization processes include lime post-stabilization, aerobic and anaerobic digestion, and composting process. The sludge produced from treating domestic and industrial wastewater generally requires stabilization before final disposal or use. Sludge lime stabilization is achieved by adding sufficient lime to sludge to raise the pH to 12 or more to disinfect microorganisms. The stabilized sludge can serve as the Class B sludge based on US standards. Aerobic digestion simply aerates the sludge in an open basin until the biodegradable substance is oxidized sufficiently to become nuisance-free. The configuration resembles the activated-sludge process, but the operation is in the endogenous respiration phase. Anaerobic digestion solubilizes and ferments complex organic substances using microorganisms in the absence of oxygen. The products of anaerobic digestion include methane, carbon dioxide, other trace gases, and the stabilized sludge. The digestion process can effectively inactivate the pathogens in the sludge. Sludge composting is a digestion process in which solid organic material undergoes biological degradation to produce a stable end product, with volatile solids converted to CO_2+H_2O. This process requires high oxygen concentration all over the composting pile to prevent creation of an anaerobic environment.

Thermal processes include drying, wet air oxidation, and others. Drying can considerably reduce sludge volume. The end product can be used as a fertilizer, soil amendment, and fuel or raw material for other thermal processes, presenting the maximum flexibility to the final disposal options. Dozens of new drying plants are being installed globally. Wet air oxidation oxidizes organic matters in an aqueous environment at temperature between 120 and 400°C, much lower than those for conventional combustion (800–1,500°C). The system pressure ranges 70–100 atms. This process produces no ash, SOx, or NOx. Moreover, no preliminary dewatering or drying is required; however, the feed concentration and heat value significantly influence operating costs. Recent progress is to use supercritical water oxidation in a deep shaft.

6. Moisture Content of Sludge

Sludge collected from a wastewater treatment plant is a suspension of 1–2% (ds). Separating water from sludge requires the input of a very large amount of energy at high cost. One useful heuristic is, therefore, "always concentrate the sludge in the treatment train." Each sludge treatment unit operates within some range of solid fractions. For instance, as shown in Fig. 6, anaerobic digestion normally operates on 3–6% solids, while landfilling requires a cake with a solid fraction of over 25% for good handlability. Each solid–liquid separation process yields a concentrated sludge or a dewatered cake with a range of solids fractions. For example, gravity thickening or dissolved air flotation (DAF) typically produces concentrated sludge at 3–6% ds. Centrifuging produces a much dryer cake, from 23% to 30% ds. Accordingly, based on the solid fraction, the thickener should precede the anaerobic digester, and the

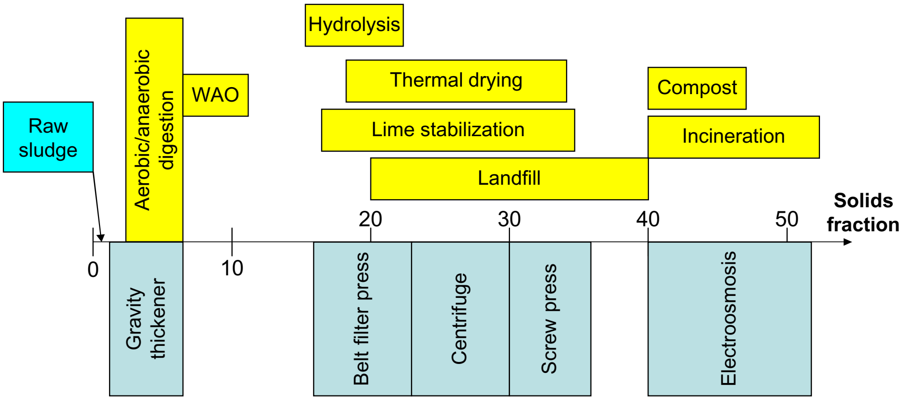


Figure 6. Solids Content Plot for the Commonly Adopted Sludge Processes and the Achievable Solid Content for Dewatering Means.

digested sludge can be further dewatered using a centrifuge before landfilling. Then, a preliminary train could be synthesized, starting from the raw sludge to the final disposal, according to the solid fraction.

To characterize the moisture content in a sludge is of main concern to process synthesis. Owing to the presence of solid phase in the sludge, certain part of water behaves very differently from the bulk water. This portion of water is commonly termed as "bound water," which resists freezing at a sub-zero temperature and is hard to be removed from the sludge body using mechanical dewatering means. Various methods have been proposed for differentiating the moisture in the sludge into bound and nonbound (free) water. The moisture could be characterized using a continuous classification scheme to provide the global information regarding the distribution status of moisture in a sludge, to give unified explanation about the bound water data obtained from different measurement techniques; and to estimate the efficiencies of various dewatering processes.

The basic idea is to estimate simultaneously the drying rate ($\overset{\bullet}{m}$, kg/s) and the heat input rate (Q, kJ/s) to a sludge body. Hence, the specific enthalpy used in moisture evaporation is evaluated by division ($\Delta H = Q/\overset{\bullet}{m}$, kJ/kg). The free water of a sludge is the portion of moisture having $\Delta H = \Delta H_{fg}$. If $\Delta H > \Delta H_{fg}$, the difference, $\Delta H - \Delta H_{fg}$, should be attributed to the existence of the solid phase of the sludge, defined as the bond strength of the moisture (H_B, kJ/kg). The moisture exhibiting greater bond strength would require more enthalpy for evaporation, hence being more difficult to be mechanically dewatered.

Figure 7 demonstrates a continuous distribution of moisture in waste-activated sludge. Interestingly, when this sludge is dewatered from a diluted suspension to a cake of around 20% solids fraction, the bond strength is close to zero. Mechanical dewatering means can easily dewater this sludge to a solid fraction of 20%. However,

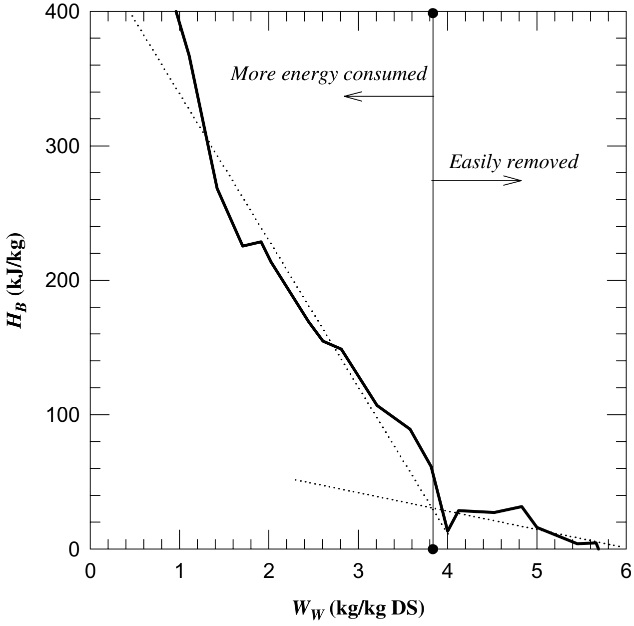


Figure 7. Continuous Distribution of Moisture in a Waste-Activated Sludge.

further dewatering requires much higher energy input, and is not achievable by using low-shear mechanical dewatering means. A high-shear centrifuge or belt filter can dewater the sludge to around 25% solid fraction. Thermal drying is normally required to dewater the sludge at a solid fraction higher than 40%.

The residual moisture content of sludge can be estimated using the continuous moisture distribution scheme with H_B as input parameter for specific dewatering means. Normally, the pressure filter provides an H_B of around 20 kJ/kg, while the expression press giving an $H_B = 60-100$ kJ/kg. Figure 8 presents the estimated moisture content of dewatered cake and the test results using an industrial press. The correlation is satisfactory, indicating that we can consider the dewatering process in a unified way by considering it as an energy supply-dissipation balancing process.

7. Synthesis of Management Systems

Any sludge management system comprises of several sub-units, each of which must be evaluated and synthesized as a part of the whole system, subject to constraints, including pollutant limits, budget, available land, local technical level, and the need for a good relationship with the public.

Figure 9 presents the synthesis route for sludge management systems. The design includes several stages. The conceptual design of a sludge treatment train begins with

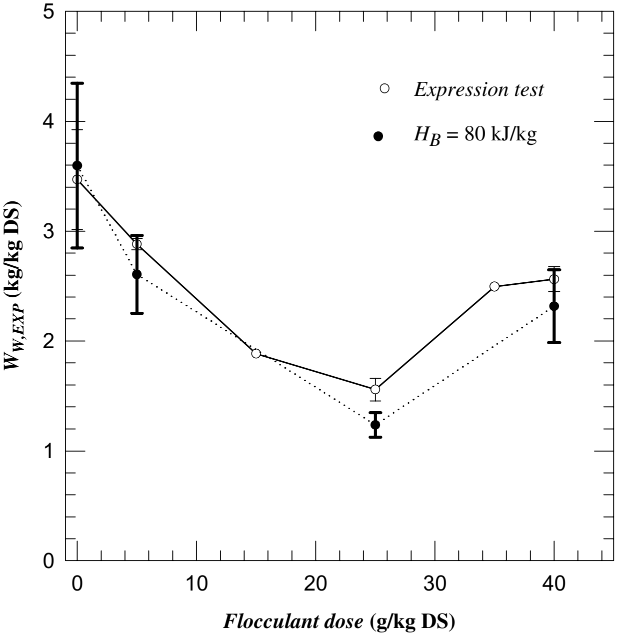


Figure 8. Residual Water Content for Sludge After Consolidation.

<table>
<tr><th colspan="2">Conceptual design</th></tr>
<tr><td colspan="2">● Sludge characterization
● Setup of final disposal
● Preliminary process synthesis
● Heat and mass balance</td></tr>
<tr><th colspan="2">Basic design</th></tr>
<tr><td colspan="2">● Treatment train synthesis
● Process integration
● Lab test
● Detailed database</td></tr>
<tr><th>Plant design</th><th>Controllability</th></tr>
<tr><td>● Equipment sizing
● Cost estimation
● Profitability analysis
● Optimization</td><td>● Control structure synthesis
● Controllability assessment
● Start-up assessment
● Dynamic simulation</td></tr>
<tr><th colspan="2">Final design</th></tr>
<tr><td colspan="2">● P&I Diagram
● Reliability and safety analyses
● Risk assessment</td></tr>
</table>

Figure 9. The Route to Synthesize Sludge Management Processes.

a little analysis. In this stage, the sludge is characterized and final disposal options are initially screened. Then, many alternative management plans are considered, based mostly on experience and heuristics. Heat and mass balances are calculated and the preliminary process scheme is defined in this stage. Thereafter, the favorable alternatives go forward to the following basic design stage. The basic design stage synthesizes the treatment train according to energy and material integration analysis. Laboratory tests are performed to collect necessary design data on the treatment units chosen, from which a detailed database is built. If, at this stage, the alternative remains feasible, satisfying all the constraints, including the space requirement, energy consumption rate, public acceptance, and others, then the design is further accessed in the detailed design stage. The detailed design stage sizes and optimizes all equipment used and ensures the controllability of the entire plant. Dynamic simulation is often needed in this stage of the design. Costs are also estimated in detail in this stage. If the process still looks promising, the final design stage constructs the P&I diagram and performs safety and risk analysis.

The sewage sludge produced in Taipei City in 2020 is considered as an example for demonstration. According to the updated municipal development plan in 2020, the amount of sewage sludge produced annually would be 75,000 tons of ds for a population of 3,000,000. The waste sewage sludge (the mixture of primary and secondary sludges) was assumed to include 72% volatile solids (vs) and a solid fraction of 2.5%. This sludge is the feedstock for the sludge management train. Figure 10 demonstrates the mass balance of sewage and sewage sludge for Taipei City in 2020.

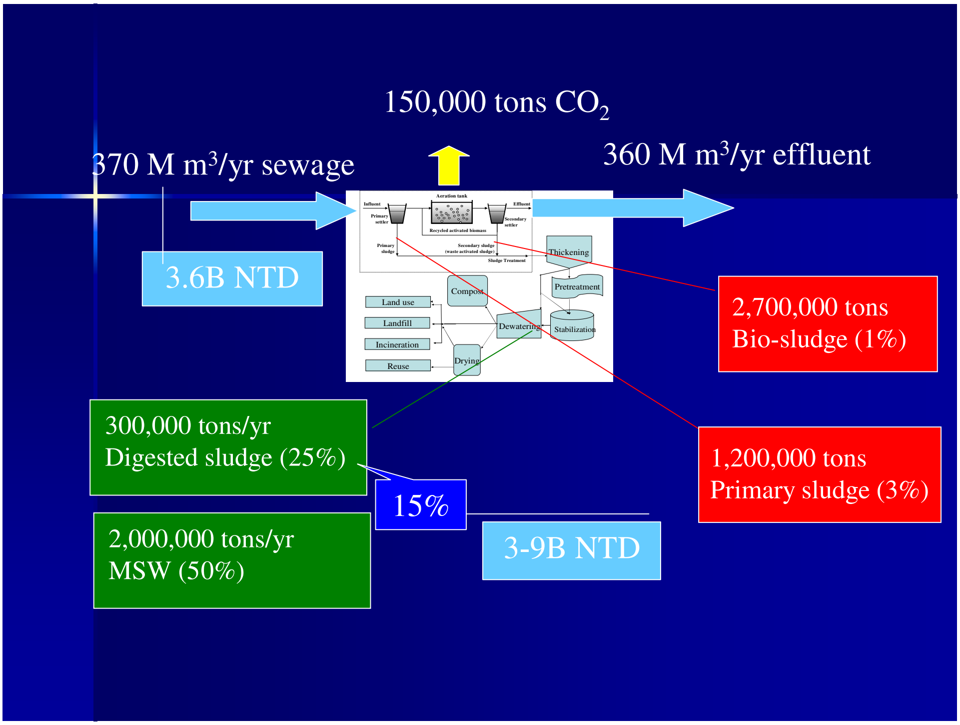


Figure 10. Mass Balance of Sewage Treatment Plant of Taipei in 2020.

The following four alternative means of disposing of Taipei's sludge have been proposed. (1) The sludge is dewatered mechanically and sent to a landfill site 50 km away from the city. (2) The sludge is incinerated using a fluidized sludge incinerator 25 km away from the wastewater treatment plant and the ash is fixed and stored. (3) The sludge is stabilized to Class A sludge to be used as fertilizer in city parks. (4) A combined solution is implemented, comprising all three of the foregoing alternatives. The management trains for all four options must be synthesized. Alternatives 1–3 depend on distinct sludge characteristics. For example, if the sludge were incinerated, the heat value of the sludge is of primary concern. The levels of heavy metals and pathogens must be sufficiently reduced before the sludge can be used on horticultural land. When landfills are used, the rheological properties and moisture content determine the handlability of the sludge; moreover, the biodegradable organic matter in the sludge is broken down before landfilling, to prevent methane production. The desired treatment process train is to convert the raw sludge efficiently, as required by the various alternatives.

Sludge collected from a wastewater treatment plant is a suspension of 1–2% ds. Separating water from sludge requires the input of a very large amount of energy at high cost. One useful heuristic is, therefore, "always concentrate the sludge in the treatment train." Each sludge treatment unit operates within some range of solid fractions. For instance, as shown in Fig. 6, anaerobic digestion normally operates on 3–6% solids, while landfilling requires a cake with a solid fraction of over 25% for good handlability. Each solid–liquid separation process yields a concentrated sludge or a dewatered cake with a range of solids fractions. For example, gravity thickening or DAF typically produces concentrated sludge at 3–6% ds. Centrifuging produces a much dryer cake, from 23% to 30% ds. Accordingly, based on the solid fraction, the thickener should precede the anaerobic digester, and the digested sludge can be further dewatered using a centrifuge before landfilling. Then, a preliminary train could be synthesized, starting from the raw sludge to the final disposal, according to the solid fraction.

Table 14 lists the preliminary treatment trains for Alternatives 1–3 for treating the sewage sludge of Taipei City. Polymer is used for centrifugal dewatering, to produce dry cake for incineration.

Table 14. Preliminary Trains Synthesized. Alt 1: Landfilling Sludge 50 km Away from the City; Alt 2: Incinerating Sludge at 25 km Away from the City; and Alt 3: Apply to Land after Composting.

	Cond.[a]	Cond.[b]	GT	BF	Centr.	Stab.[c]	Comp.	FBF	LandF	LandU	Hauling
Alt 1	X		X	X		X			X		X
Alt 2		X	X		X			X			X
Alt 3	X		X	X			X			X	X

[a]: conditioning using lime; [b]: conditioning using polymer; [c]: anaerobic digestion or lime stabilization.

Whether or not the proposed, preliminary process is feasible depends on the capital and O&M costs. Figures 11(a) and 11(b) summarize current costs of dealing with sewage sludge in Taipei City. Accordingly, the capital costs associated with landfilling sludge, with anaerobic digestion or lime stabilization as pretreatment, are US$27M and 37M, respectively. Meanwhile, the capital costs of Alternatives 2 and 3 are estimated at US$57M and US$39M, respectively. The O&M costs of the four options mentioned above are US$370, US$350, US$360, and US$180/ton-ds. These costs do not include the costs allocated to utilities, contingency costs, and costs of land, royalty, and start-up. This analysis shows that the cost of landfilling is similar to that of incineration, although the capital cost associated with the latter is higher.

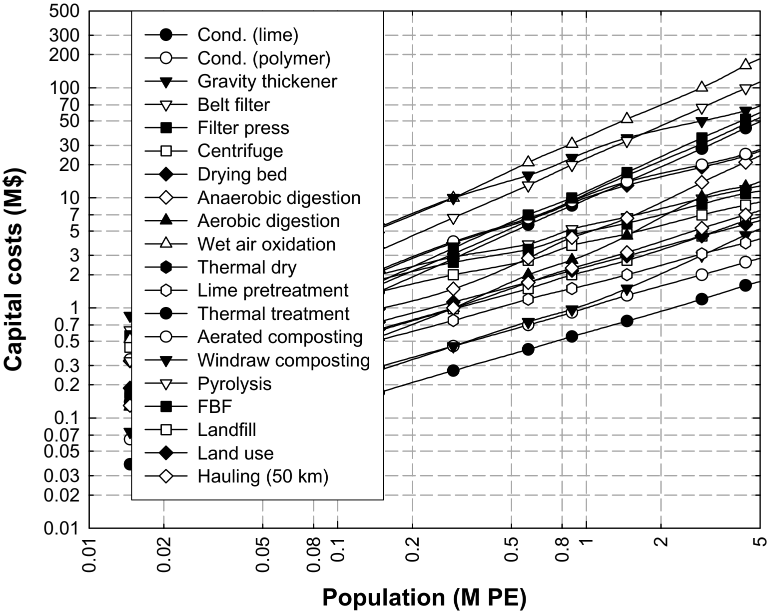
8. Sustainable Management

Peter Matthews defined "sustainable" sludge management as disposal in such a way that environmental quality and safety were restored within 50 years of disposal. "Sustainable" sludge management practices should be acceptable to the public and should be economical. An unknown sludge manager said,"If in life the only certainties are death and taxes, then surely in sludge management the only certainty is regulatory change." Normally, the regulations are revised every 3–4 years, and generally are made more stringent. This regulatory change frequently causes another major investment losses except for that on poor sludge characterization. Sustainable sludge management must consider future changes in sludge quantity and quality; moreover, it must maximize flexibility in adopted treatment processes and final disposal routes.

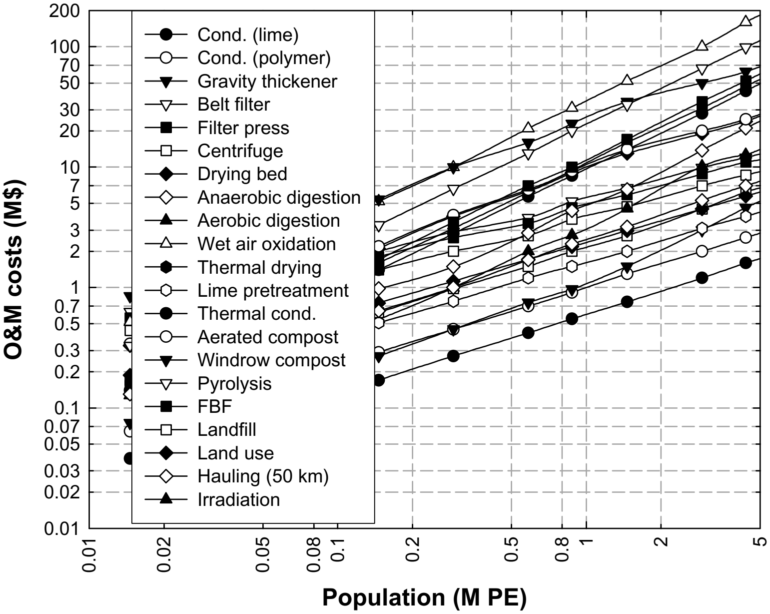
Figure 12(a) schematically illustrates conventionally used sludge disposal methods. Very clean or much polluted sludges are both rare. Sludge with contaminant concentrations exceeding the limits for agricultural use should not be disposed of on farmland. Moreover, sludge with contaminant levels exceeding the limits for hazardous waste should not be disposed of in ordinary landfill sites. The figure shows that all sludges with high contaminant levels are incinerated for final disposal.

Figure 12(b) schematically displays future sludge disposal routes. Sludge production increases owing to the new installation or upgrading of wastewater treatment facilities. Good source control on pollutants could reduce the hazardous potential of sludge. Sludge landfill should decline in line with global trends. Meanwhile, the disposal of sludge through land application should increase owing to this disposal method being more sustainable. However, regulations are likely to become increasingly strict, which will simultaneously limit sludge disposal through land application. The remaining sludge should be incinerated. Consequently, both land application and incineration amount should be increased with time.

The quantity of sludge requiring disposal will increase in the future. Any decisions made today on the sludge management system must fit the future regulatory and

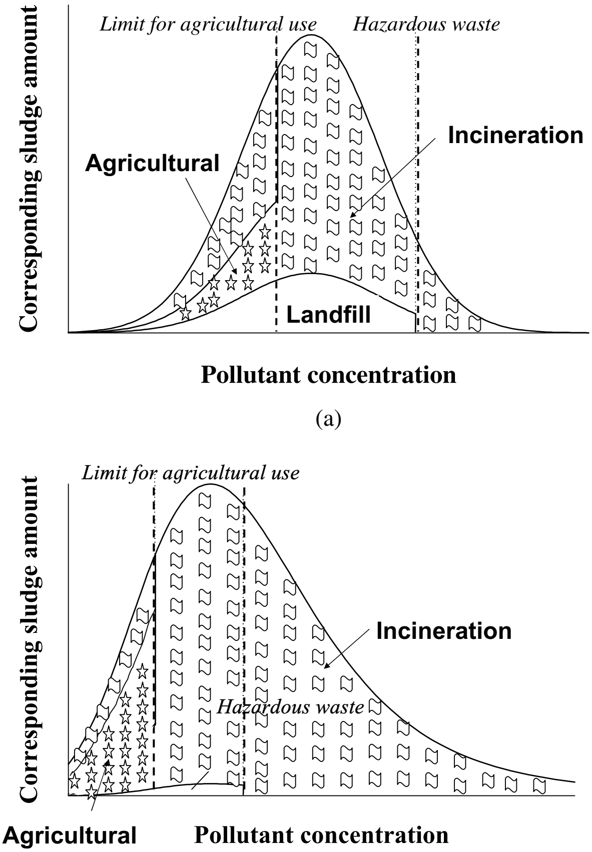


(a)



(b)

Figure 11. (a) Capital Cost for Various Treatment Units and (b) the O&M Costs for Various Treatment Units.



(a)

(b)

Figure 12. (a) The Schematics of the Sludge Disposal Routes (current practice) and (b) The Schematics of Sludge Disposal Routes (tomorrow's practice).

economic situation, which is a challenge. General roles include adopting a disposal network incorporating multiple options rather than a single option for processing and marketing, considering process site capability to accommodate increased waste streams and accept other waste, and minimizing the dependence of processing cost on fuel or chemical products.

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Chapter 3

LANDFILL FOR SOLID WASTE DISPOSAL

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Abstract

Municipal solid waste (MSW) management constitutes major environmental, economical, and social problems in the worldwide. Landfill is the most accepted method of solid waste disposal in the world. This chapter includes brief description about waste generation and waste stream to the society/community as well as minimum criteria for solid waste disposal facility siting, operations and design, groundwater monitoring and action, and closure and post-closure maintenances, etc. for landfill.

Keywords: Safety factors, operating criteria, design criteria, leachate collection system, gas emission, monitoring wells, final cover, composite liner, geomembrane and geosynthetics, glossary.

1. Introduction

1.1. *General Overview*

Waste is the most visible environmental problem among many in urban areas in the world. Human activities generate waste and the amounts tend to increase, as the demand for quality of life increases. Increasing population, changing consumption pattern, economic development, changing income structure, urbanization and industrialization as well modernization of life style resulted in the increased generation of solid waste that are diversified in types.[1] In last decade, continuous industrial and commercial growth in many countries had been accompanied by rapid increases in both the municipal and the industrial solid waste generations. Municipal solid waste (MSW) generation continues to grow in both per capita and overall terms.[2]

Figure 1 shows the total MSW and per capita waste generation in the USA from 1960 to 2006.[3,4] The per capita generation remained same since 1990; however, the total amount of MSW generation increased due to the population growth. In countries such as Malaysia, the per capita waste generation increased from 0.85 to 1.5 kg/person/day during 1995 to 2004 and approximately 98% disposal was taken place to landfill.[5] During the latter part of 1990, annual waste production ranged from 300 to 800 kg per person in the most developed countries; however, it was less than 200 kg per person in other countries.[2] Recent years, huge amount of e-waste generated in the developed countries and the projected prediction indicate that the amount will three times higher than last decade. Figure 2 illustrates the growth scenario of the e-waste generation.

The growing rate of e-waste is three times higher than other waste streams in the USA.[6,7] Some studies reported that nearly 2 million tons of used electronics, including computers and televisions, are discarded each year, which is a mounting issue for local governments; it is anticipated that a four-fold growth in e-waste in the coming years. This is true in sense that average lifespan of a computer is estimated

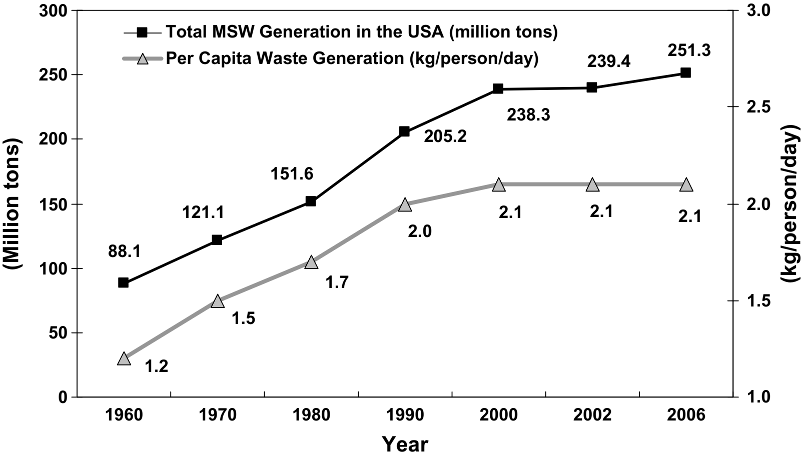


Figure 1. MSW Generation in the USA (1960–2006).[3,4]

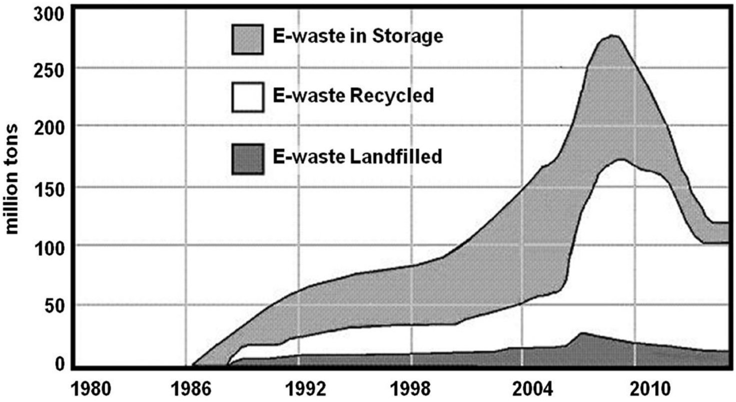


Figure 2. Projected Electronic Waste (e-waste) Scenario in the USA (1980–2010) [Some Prediction Indicated That e-waste Could Reach Three Times Higher Between 2006 and 2015].[6,7]

two years and between 1997 and 2007 more than 500 million computers became obsolete in the USA; from which more than 2% e-waste were landfilled those carried toxic heavy metals such as lead, mercury, cadmium, copper, and zinc.[8,9]

Nowadays, MSW management constitutes a major environmental, economical, and social problem worldwide, mainly due to the fact that waste volume is growing faster than the world population growth. Moreover, as stricter environmental requirements and regulations in many countries are continuously being imposed regarding ground and surface waters, therefore, observation and monitoring of landfill become major environmental concern as in most countries where landfilling is the most common way to eliminate MSW.

1.2. *Solid Waste Generation, Types and Management in a Society/ Community*

Waste includes all items that people or companies no longer have any use for which they either intended to get rid or have already discarded. Waste may take the form of solids, sludges, liquids, gases, or any combination thereof. Depending on the source of generation, some waste may degrade into harmless products, whereas other produces nondegradable and hazardous contaminants. Many items can be considered as waste, for instance household rubbish, sewage sludge, waste from manufacturing activities, packaging items, discarded cars, discarded electronic devices, garden waste, old paint containers, etc. However, the material flow in our community and/or society is illustrated in Fig. 3[10] and the waste generation and recovered components are presented in Tables 1 and 2.[11]

Finally, the consumed raw material returns to the environment as land disposal in the form of landfills, surface impoundment, land application, deep well injection, etc. The current state-of-art of solid waste disposal is based on the concept of the integrated management. The European Union regulation proposes a hierarchical based on four subsequent levels (Fig. 3): (i) reduction of the solid waste production, (ii) recovery of material, (iii) recovery of energy; and (iv) landfill disposal. There is not sufficient knowledge of this hierarchy to develop adequate disposal of waste due to the variables relating to the environmental, social, and economic aspects along with the technical aspects. Solid waste cannot responsibly be dumped without due concern and preparation, because not only it is unsightly, unhygienic, and potentially

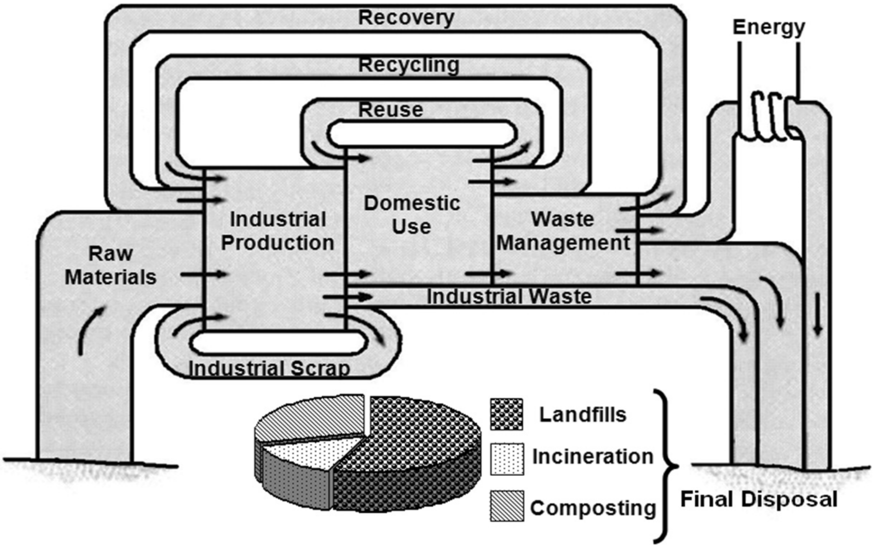


Figure 3. Schematic View of Material Flow and Waste Management in our Society.[10]

Table 1. Sources of Solid Wastes in Our Society or Community.[11]

Source	Typical Facilities, Activities, or Locations Where Waste are Generated	Type of Solid Wastes
1. Households	Single and combined house, low-, medium-, and high-rise apartment's household activities.	Food wastes, paper, cardboard, plastics, textiles, leather, yard waste, wood, glass tin cans, aluminum, others metals, ashes, street leaves, special wastes (including bulky items, e-waste, consumer electronics, white goods, yard wastes collected separately, batteries, oil, and tires), and household hazardous wastes.
2. Commercial		
(a) General	Stores, restaurants, markets, office buildings, hotels, motels, print shops, service stations, auto repair shop, etc.	Paper cardboard, plastics, wood, food waste, glass, metals, hazardous wastes, etc.
(b) Industrial	Construction, fabrication, light and heavy manufacturing, refineries, chemical plants, process plants, power plants, demolition, organic and inorganic products, etc.	Industrial process wastes, scrap materials, etc. Nonindustrial wastes including food wastes, rubbish, ashes, demolition and construction wastes, and special wastes, and hazardous wastes
(c) Infrastructure and demolition	New construction sites, road-railway repair/renovation sites, razing of buildings, and broken pavement.	Debris, rods, wood, steel, concrete, dirt, etc.
3. Institutional	Schools, hospitals, prisons, governmental centers, etc.	Paper, cardboard, food waste, glass, metals, special waste (lab waste chemicals), etc.
4. Agricultural	Field and row crops, orchards, vineyards, dairies, feedlots, farms, fertilization, chemical applications, etc.	Spoiled food wastes, agricultural crop residue and wastes, rubbish wastes, organic wastes, and inorganic hazardous wastes (excess fertilizer, pesticide, herbicide, etc.).
5. Municipal services		
(a) General cleaning	Street cleaning, landscaping, catch basin cleaning, parts and beaches, other recreational areas.	Special wastes, rubbish, street sweepings, landscape and tree trimmings, catch basin debris, general wastes from parks, beaches, and recreational areas.
(b) Treatment plant incinerators	Water, wastewater and industrial treatment processes, etc.	Treatment plant wastes, principally composed of residual sludges.
(c) Solid waste	Households and commercial and institutional activities and processes.	Paper, metals, plastics, rubber, debris, rubbish, and food.

Table 2. Materials that Have Been Recovered for Recycling From MSW.[11]

Recyclable Material	Type of Materials or Uses
Paper	Old newspaper — Newsstand and home-delivered newspapers Corrugated cardboard — Bulk packaging, the largest single source of waste paper for recycling High-grade paper — Computer paper, white ledger paper, and trim cuttings Mixed paper — Various mixtures of clean paper, including newsprint, magazines, and white and colored long-fiber paper
Plastics	Polyethylene terephthalate — Soft drink bottles, salad dressing, vegetable oil bottles, and photographic film HDPE — Milk jugs, water containers, detergent, and cooking oil bottles PVC — Home landscaping irrigation piping, some food packaging, and bottles LDPE — Thin film packaging and warps, dry cleaning bags, other film material Poly propylene — Closures and labels for bottles and containers, battery casings, bread and cheese wraps, cereal box liners Polystyrene — Packaging for electronic and electronic components, foam cups, fast-food containers, table ware, and microwave plates Multilayer and other — Multilayered packaging and ketchup and mustard bottles Mixed plastics — Various combinations of the above-mentioned products
Glass	Clean, green, and brown glass bottles and containers
Metals	Aluminum soft drink and beer cans, ferrous tin cans, white goods, and other metals, such as copper and lead
Organic and yard wastes	Used to prepare compost for soil applications, compost for use as intermediate landfill cover, methane, ethanol and other organic compounds, refuse-derived fuel, etc.
Infrastructure and demolition	Soil, asphalt, concrete, wood, drywall, shingles, metals, packing materials pallets, scraps, and used wood from construction projects
Grease and oil	Automobile, locomotive and truck oil, reprocessed for reuse or fuel
Tires	Automobile and truck tires, road building material, and fuel
Batteries	Automobile and truck batteries, shredded to recover individual components such as acid, plastic, and lead, and potential recovery of zinc, mercury, and silver from household batteries
E-waste	Copper circuits, lead recover, plastics reuse, etc.

disastrous to our environment, but also it requires the allocation of space and incurs costs related to the consequences of the waste disposal.[12,13]

1.3. *Landfilling for Solid Waste Disposal*

Landfilling has been the most economical and environmentally accepted method of solid waste disposal in the United States and in the world. Implementation of waste reduction, recycling, and transformation technologies has decreased landfill burdens but landfills remain an important component of an integrated solid waste management strategy.[11]

An MSW landfill (MSWLF) unit is defined as a discrete area of land or excavation that receives household waste, and that is not considered a land application unit, surface impoundment, and injection well, or waste pile, etc. In addition to household waste, an MSWLF unit may receive commercial waste, nonhazardous solid waste from industrial facilities including nonhazardous sludges, and sewage sludge from wastewater treatment plants.[14] The components of commercial solid waste, industrial waste, and household waste are already presented in Table 1. Typical views of landfill with essential components are illustrated in Fig. 4.[15]

Modern landfills are well-engineered facilities that are located, designed, operated, and monitored to ensure compliance with safe water and environmental regulations. Solid waste landfills must be designed to protect the environment from contaminants that may be present in the solid waste stream. The landfill siting plan that prevents the siting of landfills in environmentally sensitive areas as well as on-site environmental monitoring systems that monitor for any sign of groundwater contamination and for landfill gas provided additional safeguards. In addition, many new landfills collect potentially harmful landfill gas emissions and convert the gas into energy.

MFWLFs generally receive household waste; however, it can also receive non-hazardous sludge, industrial solid waste, and construction and demolition debris. In the USA, all MSWLFs must comply with the federal regulations of United State Environmental Protection Agency (US EPA) [*Title 40, Part 258 of the Code of Federal Regulations, Solid Waste Disposal Facility Criteria (SWDFC), commonly referred to as Subtitle D of the Resource Conservation and Recovery Act–RCRA,*

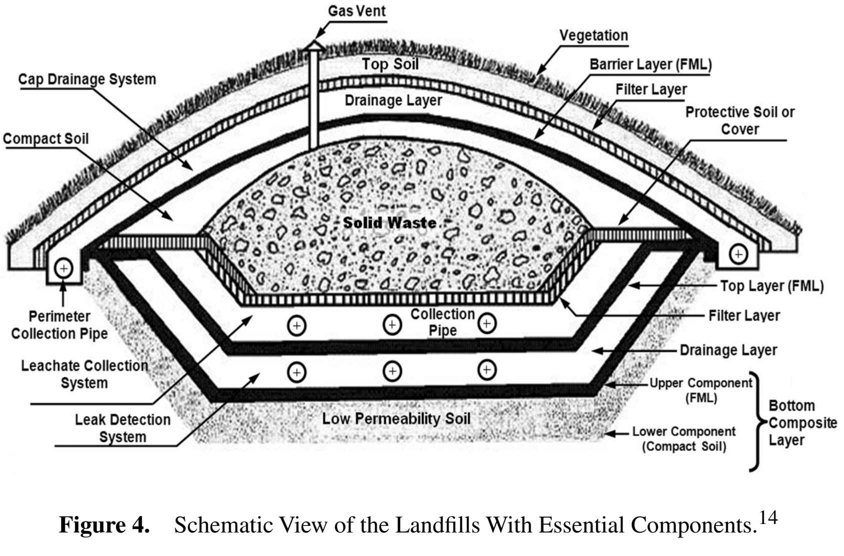


Figure 4. Schematic View of the Landfills With Essential Components.[14]

became effective on October 9, 1993], or equivalent state regulations that establish minimum criteria for solid waste disposal facility siting, design, operations, groundwater monitoring and corrective action, and closure and post-closure maintenance, etc. Federal MSWLF standards include the following considerations:

- *Location restrictions* ensure that landfills are built in suitable geological areas away from faults, wetlands, flood plains, or other restricted areas.
- *Composite liners requirements* include a flexible membrane (geomembrane) overlaying two feet of compacted clay soil lining the bottom and sides of the landfill and protect groundwater and the underlying soil from leachate releases.
- *Leachate collection and removal systems* sit on top of the composite liner and remove leachate from the landfill for treatment and disposal.
- *Operating practices* include compacting and covering waste frequently with several inches of soil help reduce odor; control litter, insects, and rodents; and protect public health.
- *Groundwater monitoring requirements* require testing groundwater wells to determine whether or not waste materials have escaped from the landfill.
- *Closure and post-closure care requirements* include covering landfills and providing long-term care of closed landfills.
- *Corrective action provisions* control and clean up landfill releases and achieve groundwater protection standards (GWPSs).
- *Financial assurance* provides funding for environmental protection during and after landfill closure (i.e., closure and post-closure cares).

Except the financial assurance, these standard considerations are described in the subsequent sections as (1) location criteria, (2) operating criteria, (3) design criteria, (4) groundwater monitoring and corrective action, and (5) closure and post-closure care.

2. Location Criteria

2.1. *General Overview*

For newly constructed MSWLF or lateral expansion or existing MSWLF followed some location restriction that addresses both the potential effects that an MSWLF unit may have on the surrounding environment and the effects that natural and human-made conditions may have on the performance of the landfill unit. The location criteria covered the following:

- airport safety,
- floodplains,

- wetlands,
- fault areas,
- seismic impact zones, and
- unstable areas.

Floodplain, fault area, seismic impact zone, and unstable area restrictions address conditions that may have adverse effects on landfill performance that could lead to releases to the environment or disruptions of natural functions (i.e. floodplain flow restrictions). Airport safety, floodplain, and wetland criteria are intended to restrict MSWLF units in areas where sensitive natural environments and/or the public may be adversely affected. In Table 3,[11,16,17,19] the conditions of location restrictions are presented briefly and Table 4[19] provides a quick reference to the location standards required by the criteria.[11,16,17,19]

Table 3. Location Limitations for MSWLF Constructions/Operations in the Regulation of US EPA.[11,16,17,19]

Locations	Location Limitations
Airport	10,000 ft from an airport where used by turbojet aircraft; 5,000 ft from an airport where used by piston type aircraft. Any landfills closer will have to demonstrate that they do not pose a bird hazard to aircraft.
Flood plains	100-year flood plain. Landfill located within the 100 year floodplain will have to be designed so as not to restrict flood flow, reduce the temporary water storage capacity of the flood plain, or result in washout of the solid waste, which would pose a hazard to human and the environment.
Wetlands	New landfills will not be able to locate in wetlands unless the following conditions have been demonstrated: (1) no practical alternative with less environmental risk exists; (2) violations of other state and local laws will not occur; (3) the unit would not cause or contribute to significant degradation of the wetland; (4) appropriate and practicable steps have been taken to minimize potential adverse impacts; and (5) sufficient information to make determination is available.
Fault areas	New landfill units cannot be sited within 200 ft of a fault line that has had a displacement in Holocene time (past 10,000 years).
Seismic impact zone	New landfill unit located within seismic impact zone will have to demonstrate that all contaminant structures (liners, LCSs, and surface water control structures) are designed to resist the maximum horizontal acceleration in lithified materials (liquid or loose materials consolidated into solid rock) for the site.
Unstable areas	Landfill units located in unstable areas must demonstrate that the design ensures stability of structure components. The unstable areas include the areas that are landslide prone, that are in karst geology susceptible to sinkhole formation, and that are undermined by subsurface mines. Existing facilities that cannot demonstrate the stability of the structural components will be required to close within five years of the regulation's effective date.

Table 4. Location Criteria Standard (US EPA 40 CFR — Title 40: Protection of Environment).[19]

Restricted Location	Applies to Existing Units	Applies to New Units and Lateral Expansions	Make Demonstration to "Direction of An Approved State" or Demonstration in Operating Record	Existing Units Must Demonstration cannot be Made
Airport	Yes	Yes	Operating record	Yes
Floodplains	Yes	Yes	Operating record	Yes
Wetlands	No	Yes	Director	N/A
Fault areas	No	Yes	Director	N/A
Seismic impact zones	No	Yes	Director	N/A
Unstable areas	Yes	Yes	Operating record	Yes

2.2. *Airport Safety*

Three major elements are considered for airport safety aspects are:

(i) MSWLF units located within 10,000 ft of any airport runway end used by turbojet aircraft or within 5,000 ft of any airport runway end used only by piston-type aircraft.
(ii) Airports runways are open to the public without prior permission for use, and where the use of available facilities is not restricted, and
(iii) Potential probability about bird/aircraft collisions that may causes damage to the aircraft and injury to its occupants.

If the above conditions are present, the MSWLF owner or operator must demonstrate that the MSWLF unit does not pose a bird hazard to aircraft and notify the concern authorities that the demonstration has been placed in the operating record. Otherwise, the existing units must be closed.

The first element can be addressed by using existing maps and the measurement can be made by drawing a circle of appropriate radius (i.e. 10,000 ft or 5,000 ft) depending on airport type. The measurement should be considered between the end of the runway and the nearest MSWLF unit perimeter, not between any other boundaries. The second elements can be addressed to determine whether or not the runway is part of a public use airport and to determine whether or not all applicable public airports have been identified, the MSWLF unit owner/operator should contact the airport authority; however, this rule does not apply to any private airfields.

The MSWLF unit design features and operational practices can have a significant effect on the likelihood of increased bird/aircraft collisions. Birds may be attracted to MSWLF units to satisfy a need for water, food, nesting, or roosting. Scavenger birds such as starlings, crows, blackbirds, and gulls are most commonly associated with active landfill units. Where bird/aircraft collisions occur, these types of birds are often involved due to their flocking, feeding, roosting, and flight behaviors. Waste management techniques to reduce the supply of food to these birds include:

- Frequent covering of wastes that provide a source of food;
- Shredding, milling, or baling the waste-containing food sources; and
- Eliminating the acceptance of wastes at the landfill unit that represent a food source for birds (by alternative waste management techniques such as source separation and composting or waste minimization).

Frequent covering of wastes that represent a food source for the birds effectively reduces the availability of the food supply. By maintaining a small working face, spreading and compaction equipment are concentrated in a small area that further disrupts scavenging by the birds. The use of varying bird control techniques may prevent the birds from adjusting to a single method. Methods such as visual deterrents or sound have been used with mixed success in an attempt to discourage birds from food scavenging. Visual deterrents include realistic models (still or animated) of the bird's natural predators (i.e. humans, owls, hawks, and falcons). Use of physical barriers such as fine wires strung across or near the working face have also been successfully used (Fig. 5).[19]

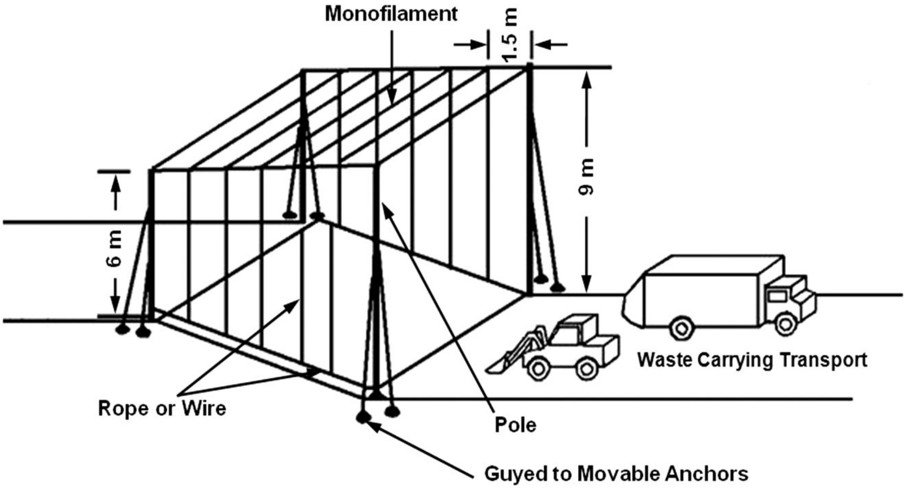


Figure 5. Illustration of a Bird Control Device.[19]

Birds also may be attracted to a landfill unit as a nesting area. Use of the landfill site as a roosting or nesting area is usually limited to ground-roosting birds (i.e. gulls). Operational landfill units that do not operate continuously often provide a unique roosting habitat due to elevated ground temperatures (as a result of waste decomposition within the landfill) and freedom from disturbance. Nesting can be minimized, however, by examining the nesting patterns and requirements of undesirable birds and designing controls accordingly. In addition to design features and operational procedures to control bird populations, the demonstration should address the likelihood that the MSWLF unit may increase bird/aircraft collisions. One approach to addressing this part of the airport safety criterion is to evaluate the attraction of birds to the MSWLF unit and determine whether or not this increased population would be expected to result in a discernible increase in bird/aircraft collisions. The evaluation of bird attraction can be based on field observations at existing facilities where similar geographic location, design features, and operational procedures are present. All observations, measurements, data, calculations and analyses, and evaluations should be documented and included in the demonstration.

2.3. *Floodplains*

An MSWLF unit could be affected by the flow and temporary storage capacity of a floodplain. Higher flood levels and greater flood damage both upstream and downstream can be created and could cause a potential hazard to human health and safety. Therefore, new MSWLF units, existing MSWLF units, and lateral expansions of existing units that are located within a 100-year river floodplain and cannot demonstrate that the units will neither restrict the flow of a 100-year flood nor reduce the water storage capacity, and will not result in a washout of solid waste, must closed. The rule does not prohibit locating an MSWLF unit in a 100-year floodplain; where the owner or operator is allowed to demonstrate that the unit will comply with the flow restriction, temporary storage, and washout provisions of the floodplain criterion and regulation. If a demonstration can be made that the landfill unit will not pose threats, the demonstration must be placed in the operating record, and the concerned authorities must be notified that the demonstration was made and placed in the record, unless the MSWLF unit must be closed within five years and the owner or operator must conduct post-closure activities. The closure deadline may be extended for up to two years due to no available alternative disposal capacity exists and there is no immediate threat to human health and the environment. In Fig. 6[20] showed is an illustration of the potential risk of flooding zones for an MSWLF based on 200-year flooding data base by using geographical information system (GIS) modeling.

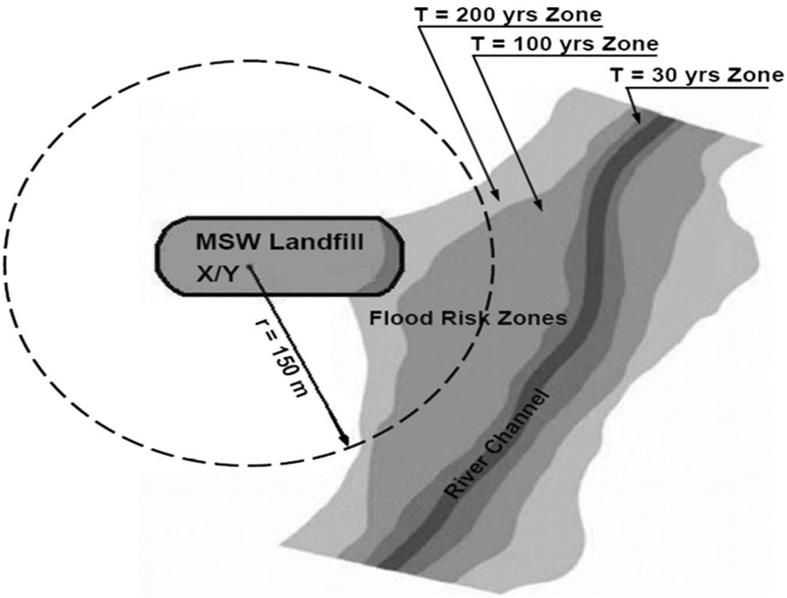


Figure 6. Schematic Illustration of the Potential Risk of Flooding Zones for an MSWLF Based on 200-year Flooding Database by Using GIS Modeling (T is Indicating the Highest Frequency of Flooding Zone Within 200, 100, and 30 years, respectively).[20]

2.3.1. *Floodplain Identification*

River floodplains are readily identifiable as the flat areas adjacent to the river's normal channel. One hundred-year floodplains represent that the sedimentary deposits were formed by floods that have a 1% chance of occurrence in any given year and that were identified in the flood insurance rate maps (FIRMs) and flood boundary and floodway maps that normally published by the National/Federal Disaster Management Agency (N/FDMA) or national meteorological department in any country or regional aspects. In Fig. 6, the areas classified as "T = 200 yrs" zones are subjected to the floodplain location restriction. Similarly, the areas classified as "T = 100 yrs" are also subjected to the floodplain location restriction where MSWLF are presented within 150-m radius. Whereas "T = 30 yrs" zones are not subjected to the restriction, although care should be taken to design facilities capable of withstanding some potential flooding.

Many of the river channels were covered by FIRMs maps may have undergone modification for hydropower or flood control projects and, therefore, the floodplain boundaries represented may not be accurate or representative. It may be necessary to compare the floodplain map series to recent air photographs and satellite images to identify current river channel modifications and land use watersheds that could affect floodplain designations. If floodplain maps are not available and the facility is located within a floodplain, then a field study to delineate the 100-year floodplain may be required as well as can query by using GIS data sets. A floodplain delineation program can be based primarily on meteorological records and physiographic information such as existing and planned watershed land use, topography, soil and geologic mappings, and air photo interpretation of geomorphologic (i.e. land formation) features. In the USA, Water Resource Council (WRC) provides information for determining the potential for floods in a given location by stream gauge records and GIS modeling. Estimation of the peak discharge also allows an estimation of the probability of exceeding the 100-year flood.

2.3.2. *Engineering Considerations*

If the MSWLF unit is within the 100-year floodplain, then the MSWLF unit does not significantly restrict the base flood flow or significantly reduce temporary storage capacity of the floodplain. Therefore, the MSWLF unit must be designed to prevent the washout of solid waste during the expected flood event. The rule requires that floodplain storage capacity and flow restrictions, which occur because the MSWLF unit, do not pose a hazard to human health and the environment. The demonstration that these considerations are met relies on the estimates of the flow velocity and volume of floodplain storage in the vicinity of the MSWLF unit during the base flood. The assessment should consider the floodplain storage capacity and floodwater velocities that would likely exist in the absence of the MSWLF unit. The volume occupied by an MSWLF unit in a floodplain may theoretically reduce the storage capacity and restrict flow. Raising the base flood level by more than one foot can be an indication that the MSWLF unit may reduce and restrict the storage capacity flow. The location of the MSWLF unit relative to the velocity distribution of floodwaters will greatly influence the susceptibility to washout. This type of assessment will require a conservative estimate of the shear stress on the landfill components caused by the depth, velocity, and duration of impinging river waters. Depending on the amount of inundation, the landfill unit may act as a channel side slope or bank or it may be isolated as an island within the overbank river channel. In both cases, an estimate of the river velocity would be part of a proper assessment. The assessment of flood water velocity requires that the channel cross section be known above and below the landfill unit. Friction factors on the overbank are determined from the surface conditions and vegetation present. River hydrologic models may be used to simulate flow levels and estimate velocities through these river cross sections.

In recent years, many countries have developed numerical models, GIS and satellite-image-based models, to aid in the prediction of flood hydrographs, flow parameters, the effect of obstructions on flow levels, the simulation of flood control structures, and sediment transport, etc., which are effectively used for floodplain modeling in the world wide.

2.4. *Wetlands*

Wetlands are areas that are inundated or saturated by surface water or groundwater at a frequency and duration sufficient to support and that under normal circumstances do support a prevalence of vegetation typically adapted for life in saturated soil conditions.[27] It includes swamps, marshes, bogs, and any areas that are inundated or saturated by groundwater or surface water as defined under current guidelines; however, wetlands are identified based on the presence of hydric soils, hydrophytic vegetation, and the wetland hydrology. These characteristics also affect the functional value of a wetland in terms of its role in supporting fish and wildlife habitats; providing esthetic, scenic, and recreational values; accommodating flood storage; sustaining aquatic diversity; and its relationships to surrounding natural areas through nutrient retention and productivity exportation (i.e. releasing nutrients to downstream areas, providing transportable food sources). Often, a wetland assessment will need to be conducted by an experienced multi-disciplinary team. The assessment should identify: (1) the limits of the wetland boundary based on hydrology, soil types, and plant types; (2) the type and relative abundance of vegetation, including trees; and (3) rare, endangered, or otherwise protected species and their habitats (if any).

New landfills and lateral expansion will not be able to locate in wetlands unless the following conditions have been demonstrated: (1) no practical alternative with less environmental risk exists, (2) violations law will not occur, (3) the unit would not cause or contribute to significant degradation of the wetland, (4) appropriate and practicable steps have been taken to minimize potential adverse impacts, and (5) sufficient information to make determination is available. Moreover, there will be an objective of no net loss of wetlands in terms of acreage and function; it recognizes that the regions of any country where proportionally large areas are dominated by wetlands. In these regions, sufficient acreage and a suitable type of upland may not be present to allow construction of a new MSWLF unit or lateral expansion without wetland impacts. Wetlands evaluations may become an integral part of the siting, design, permitting, and environmental monitoring aspects of a landfill unit/facility (Fig. 7).[19]

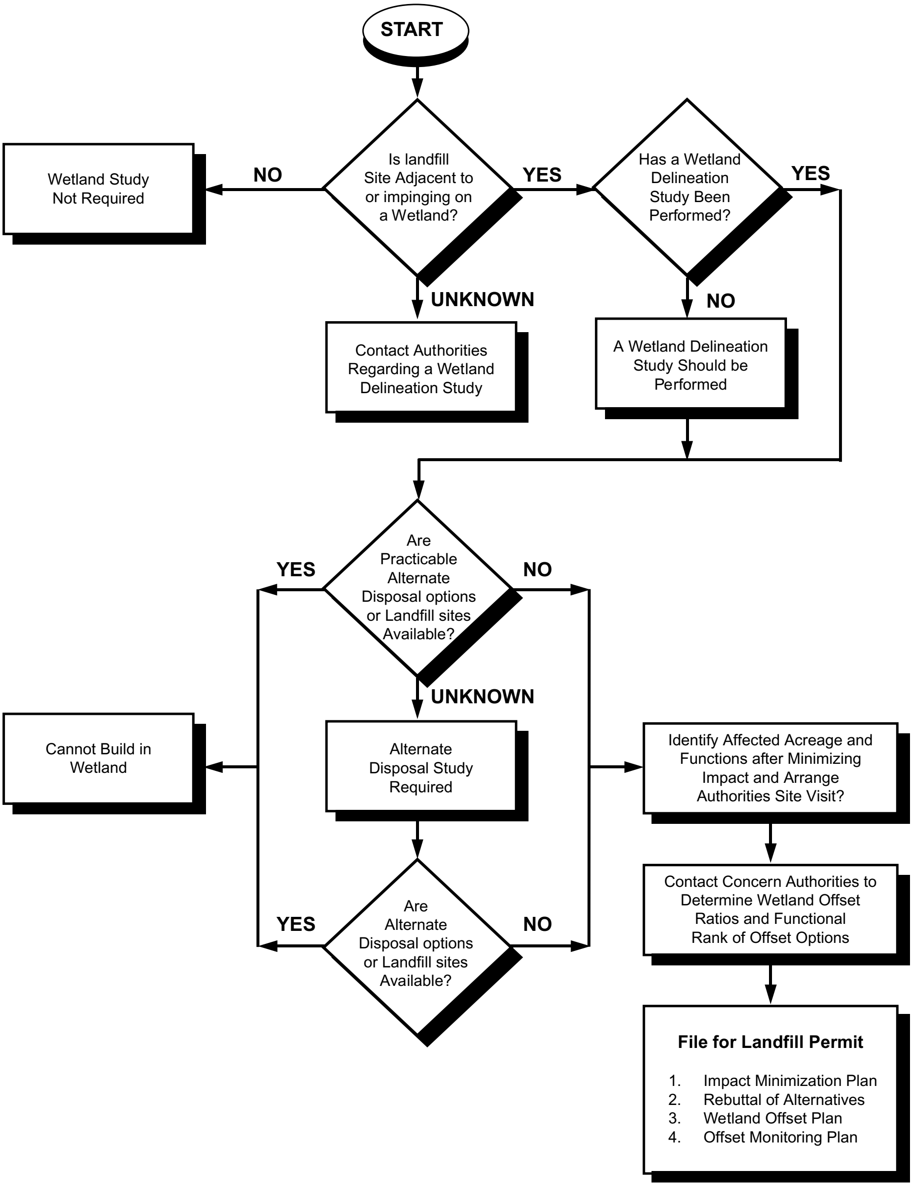


Figure 7. Schematic Illustration of Wetland Decision Tree for Landfill Owner/operators.[19]

The owner or operator must show that the operation or construction of the landfill unit will not:

- violate any applicable water quality standards;
- cause or contribute to the violation of any applicable toxic effluent standard or prohibition;

- cause or contribute to violation of any requirement for the protection of a marine sanctuary; and
- jeopardize the continued existence of endangered or threatened species or critical habitats.

The MSWLF unit cannot cause or contribute to significant degradation of wetlands. Therefore, the owner/operator must:

- ensure the integrity of the MSWLF unit, including consideration of the erosion, stability, and migration of native wetland soils and dredged/fill materials;
- minimize impacts on fish, wildlife, and other aquatic resources and their habitat from the release of solid waste;
- evaluate the effects of catastrophic release of wastes on the wetlands; and
- assure that ecological resources in the wetlands are sufficiently protected, including consideration of the volume and chemical nature of waste managed in the MSWLF unit.

After consideration of these factors, if no practicable alternative to locate the landfill in wetlands is available, compensatory steps must be taken to achieve no net loss of wetlands as defined by acreage and function. The owner/operator must try to avoid and/or minimize impacts to the wetlands to the greatest extent possible. Where avoidance and minimization still result in wetland impacts, mitigation to offset impacts is required. Mitigation plans must be approved by the appropriate regulatory agencies and must achieve an agreed-upon measure of success. Examples of mitigation include restoration of degraded wetlands or creation of wetland acreage from existing uplands. Moreover, practicable alternatives are available to locate landfill units in wetlands because landfilling is not a water-dependent activity. The term "practicable" pertains to the economic and social feasibilities of alternatives (i.e., collection of waste at transfer stations and trucking to an existing landfill facility or other possible landfill sites). The feasibility evaluation may entail financial, economic, administrative, and public acceptability analyses as well as engineering considerations. Furthermore, the evaluations generally will require generation and assessment of land use, geologic, hydrologic, geographic, demographic, zoning, and traffic maps and other related information. To rebut the presumption that an alternative practicable site exists generally will require that a site search for an alternative location be conducted. There are no standard methods for conducting site searches due to the variability of the number and hierarchy of screening criteria that may be applied in a specific case. Typical criteria may include:

- Distance from waste generation sources;
- Minimum landfill facility size requirements;
- Soil conditions;

- Proximity to groundwater users;
- Proximity of significant aquifers;
- Exclusions from protected natural areas;
- Degree of difficulty to remediate features; and
- Setbacks from roadways and residences.

2.5. *Fault Areas*

Locating a landfill in the vicinity of an area that has experienced faulting in recent time has inherent dangers. Faulting occurs in areas where the geologic stresses exceed a geologic material's ability to withstand those stresses. Such areas also tend to be subject to earthquakes and ground failures (i.e. landslides and soil liquefaction) associated with seismic activity. Proximity to a fault can cause damage through:

- Movement along the fault that can cause displacement of facility structures,
- Seismic activity associated with faulting that can cause damage to facility structures through vibratory action, and
- Earth shaking that can cause ground failures such as slope failures.

Consequently, appropriate setbacks from fault areas are required to minimize the potential for damage. Geological survey mapping can be used to determine a proposed landfill unit is located in any Holocene fault area or not. For locations where a fault zone has been subjected, a geologic reconnaissance of the site and surrounding areas may be required to map fault traces and to determine the faults along which movement has occurred in Holocene time. This reconnaissance also may be necessary to support a demonstration for a setback of less than 200 ft. Additional requirements may need to be met before a new unit or lateral expansion may be approved. A site fault characterization is necessary to determine that a site is within 200 ft of a fault or not. An investigation would include obtaining information on any lineaments (linear features) that suggest the presence of faults within a 3,000 ft radius of the site. The information could be based on:

- A review of available maps, logs, reports, scientific literature, or insurance claim reports;
- An aerial reconnaissance of the area within a 5-mile radius of the site, including aerial photo analysis; or
- A field reconnaissance that includes walking portions of the area within 3,000 ft of the unit.

If the site fault characterization indicates that a fault or a set of faults is situated within 3,000 ft of the proposed unit, investigations should be conducted to determine the presence or absence of any faults within 200 ft of the site that have experienced movement during the Holocene period. Such investigations can include:

- Subsurface exploration, including drilling and trenching, to locate fault zones and evidence of faulting.
- Trenching perpendicular to any faults or lineaments within 200 ft of the unit.
- Determination of the age of any displacements; for example, by examining displacement of surficial deposits such as glacial or older deposits (if Holocene deposits are absent).
- Examination of seismic epicenter information to look for indications of recent movement or activity along structures in a given area.
- Review of high altitude, high resolution aerial photographs, and satellite images with stereo-vision coverage.

Based on this information as well as supporting maps and analyses, an expert should prepare a report that delineates the location of the Holocene fault(s) and the associated 200-ft setback. If requesting an alternate setback, a demonstration must be made to show that no damage to the landfill's structural integrity will result. Examples of engineering considerations and modifications that may be included in such demonstrations are as follows:

- For zones with high probabilities of high accelerations (horizontal) within the moderate range of 0.1–0.75 g, seismic designs should be developed.
- Seismic stability analysis of landfill slopes should be performed to guide selection of materials and gradients for slopes.
- Where *in-situ* and laboratory tests indicate that a potential landfill site is susceptible to liquefaction, ground improvement measures such as grouting, dewatering, heavy tamping, and excavation should be implemented.
- Engineering options include:
    - — flexible pipes,
    - — ground improvement measures (grouting, dewatering, heavy tamping, and excavation), and/or
    - — redundant precautionary measures (secondary containment system).

In addition, the use of such measures needs to be demonstrated to be protective of human health and the environment. The types of engineering controls described above are similar to those that would be employed in areas that are likely to experience earthquakes.

2.6. *Seismic Impact Zones*

A seismic impact zone is an area that has a 10% or greater probability that the maximum expected horizontal acceleration in lithified earth material, expressed as a percentage of the earth's gravitational pull (g), will exceed 0.10 g in 250 years.

New MSWLF units and lateral expansions in seismic impact zones are prohibited, unless the owner or operator can demonstrate that the structural components of the unit (e.g., liners, leachate collection systems (LCSs), final cover, and surface water control systems) are designed to resist the maximum horizontal acceleration in lithified earth material at the site. Existing units are not required to be retrofitted. Owners or operators of new units or lateral expansions must notify the concern authorities and place the demonstration of compliance with the conditions of the restriction in the operating record. A brief introduction to the geologic underpinnings of seismic activity is presented below.

The earth's crust is not a static system. It consists of an assemblage of earthen masses that are in slow motion. As new crust is generated from within the earth, old edges of crust collide with one another, thereby causing stress. The weaker edge is forced to move beneath the stronger edge back into the earth. The dynamic conditions of the earth's crust can be manifested as shaking ground (i.e. seismic activity), fracturing (i.e. faulting), and volcanic eruptions. Seismic activity also can result in types of ground failure. Landslides and mass movements (i.e. slope failures) are common on slopes; soil compaction or ground subsidence tends to occur in unconsolidated valley sediments; and liquefaction of soils tends to happen in areas where sandy or silty soils that are saturated and loosely compacted become in effect, liquefied (i.e. like quicksand) due to the motion. To determine the maximum horizontal acceleration of the lithified earth material for the site, owners or operators of MSWLF units should review the seismic 250-year interval maps. A schematic diagram of a landfill showing potential deformation of the leachate collection and removal system by using seismic stresses is presented in Fig. 8.[19]

Numerous studies indicated that during earthquakes, superficial (shallow) slides and differential displacement tend to be produced, rather than massive slope failures. Stresses created by superficial failures can affect the liner and final cover systems as well as the leachate and gas collection and removal systems of MSWLF. Tensional stresses within the liner system can result in fracturing of the soil liner and/or tearing of the flexible membrane liner (FML). Thus, when selecting suitable sites from many potential sites during the siting process, the owner/operator should try to avoid a site with:

- Holocene fault zones,
- sites with potential ground motion, and
- sites with liquefaction potential.

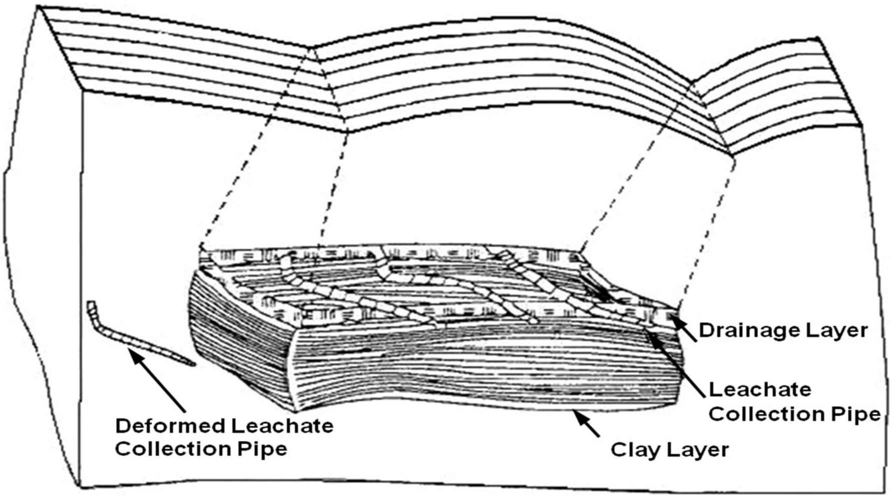


Figure 8. A Schematic Diagram of a Landfill Showing Potential Deformation of the Leachate Collection and Removal System by Using Seismic Stresses.[19]

If one of the above types of sites is selected, the owner/operator must consider the costs associated with the development of the site. If, due to a lack of suitable alternatives, a site is chosen that is located in a seismic impact zone, a demonstration must be made that the design of the unit's structural components (i.e. liners, leachate collection, final covers, and run-on and run-off systems) will resist the maximum horizontal acceleration in lithified materials at the site. As part of the demonstration, owner/operators must:

- determine the expected peak ground acceleration from a maximum strength earthquake that could occur in the area,
- determine the site-specific seismic hazards such as soil settlement, and
- design the facility to withstand the expected peak ground acceleration.

For those MSWLF units located in an area with an estimated maximum horizontal acceleration >0.1 g, an evaluation of seismic effects should consider both foundation soil stability and waste stability under seismic loading. Conditions that may be considered for the evaluation include the construction phase (maximum open excavation depth of new cell adjacent to an existing unit), closure activities (prior to final consolidation of both waste and subsoil), and post-closure care (after final consolidation of both waste and foundation soil). If the maximum horizontal acceleration is ≤ 0.1 g, then the design of the unit will not have to incorporate an evaluation of seismic effects unless the facility will be situated in an area with low strength foundation soils or soils with potential for liquefaction. The facility should be assessed for the effects of seismic activity even if the horizontal acceleration is expected to be <0.1 g.

In determining the potential effects of seismic activity on a structure, an engineering evaluation should examine soil behavior with respect to earthquake intensity. When evaluating soil characteristics, it is necessary to know the soil strength as well as the magnitude or intensity of the earthquake in terms of peak acceleration. Other soil characteristics, including degree of compaction, sorting (i.e. organization of the soil particles), and saturation, may need to be considered because of their potential influence on site conditions.

Well-compacted cohesionless embankments or reasonably flat slopes in insensitive clay are less likely to fail under moderate seismic shocks (up to 0.15–0.20 g acceleration). Embankments made of insensitive cohesive soils founded on cohesive soils or rock may withstand even greater seismic shocks. For earthen embankments in seismic regions, designs with internal drainage and core material that is most resistant to fracturing should be considered. Slope materials vulnerable to earthquake shocks are described below:

- Very steep slopes of weak, fractured, and brittle rocks or unsaturated loess are vulnerable to transient shocks caused by tensional faulting;
- Loess and saturated sand may be liquefied by seismic shocks causing the sudden collapse of structures and flow slides;
- Similar effects are possible in sensitive cohesive soils when natural moisture exceeds the soil's liquid limit; and
- Dry cohesionless material on a slope at an angle of repose will respond to seismic shock by shallow sloughing and slight flattening of the slope.

In general, loess, deltaic soils, floodplain soils, and loose fills are highly susceptible to liquefaction under saturated conditions. Geotechnical stability investigations frequently incorporate the use of computer models to reduce the computational time of well-established analytical and computer simulation methods. A conservative approach would incorporate both vertical and horizontal forces caused by bedrock acceleration if it can be shown that the types of material of interest are susceptible to the vertical force component. Typically, the horizontal force caused by bedrock acceleration is the major force to be considered in the seismic stability analysis.

Design modifications to accommodate an earthquake may include shallower waste side-slopes, more conservative design of dikes and run-off controls, and additional contingencies for leachate collection should primary systems be disrupted. Strengths of the landfill components should be able to withstand these additional forces with an acceptable factor of safety (FS). The use of professionals experienced in seismic analysis is strongly recommended for design of facilities located in the areas of high seismic risk.

2.7. *Unstable Areas*

Unstable area means a location that is susceptible to natural or human-induced events or forces capable of impairing the integrity of some or all of the MSWLF structural components responsible for preventing releases from a landfill. Unstable areas can include poor foundation conditions, areas susceptible to mass movements, and karst terrains. In this particular case, structural components for an MSWLF consist of liners, LCSs, final cover systems, run-on and run-off control systems, and any other component necessary for protection of human health and the environment.

Owners or operators of new MSWLF units, existing MSWLF units, and lateral expansions located in an unstable area must demonstrate that engineering measures have been incorporated into the MSWLF unit's design to ensure that the integrity of the structural components of the MSWLF unit will not be disrupted. The owner or operator must consider the following factors, at a minimum, when determining whether an area is unstable:

- On-site or local soil conditions that may result in significant differential settling;
- On-site or local geologic or geomorphologic features; and
- On-site or local human-made features or events (both surface and subsurface).

In the USA, existing MSWLF unit that does not meet the demonstration must be closed within five years and must undertake post-closure activities; however, it can extent another two years to the closure requirements under two conditions:

(1) no disposal alternative is available and
(2) no immediate threat is posed to human health and the environment.

Natural unstable area, including those areas that have poor soils for foundations, are susceptible to mass movement or have karst features. In brief, discussions about unstable conditions are presented as follow:

- *Poor foundation* — Areas with soils that make poor foundations have soils that are expansive or settle suddenly. Such areas may lose their ability to support a foundation when subjected to natural (i.e. heavy rain) or human-made events (i.e. explosions).
  - — Expansive soils usually are clay-rich soils that, because of their molecular structure, tend to swell and shrink by taking up and releasing water and thus are sensitive to a variable hydrologic regime. Such soils included smectite which is rich in clay (i.e. montmorillonite clay group, vermiculite clays, and bentonite). In addition, soils rich in "white alkali" (i.e. sodium sulfate), anhydrite (i.e. calcium sulfate), or pyrite (i.e. iron sulfide) also may exhibit swelling as water content increases. Such soils tend to be found in the arid areas.

  - Soils that are subject to rapid settlement (i.e. subsidence) include loess, unconsolidated clays, and wetland soils. Loess is wind-deposited silt that is moisture-deficient and tends to compact upon wetting easily. Unconsolidated clays that can undergo considerable compaction when fluids such as water or oil are removed. Similarly, wetland soils, which by their nature are water bearing, also tend to be subjected to subsidence when water is withdrawn.

- *Mass movement* — another type of unstable area which is subjected to mass movement. Such areas can be situated on steep or gradual slopes. They tend to have rock or soil conditions that are conducive to down-slope movement of soil, rock, and/or debris (either alone or mixed with water) under the influence of gravity. Examples of mass movements include avalanches, landslides, debris slides and flows, and rock slides.
- *Karst terrains* — which tend to be subjected to extreme incidents of differential settlement, namely complete ground collapse. Karst is a term used to describe areas that are underlain by soluble bedrock, such as limestone, where solution of the rock by water creates subterranean drainage systems that may include areas of rock collapse. These areas tend to be characterized by large subterranean and surficial voids (i.e. caverns and sinkholes) and unpredictable surface and groundwater flow (i.e. sinking streams and large springs). Other rocks such as dolomite or gypsum also may be subject to solution effects.

Some human-induced unstable areas are described below:

- The presence of cut and/or fill slopes during construction of the MSWLF unit may cause slippage of existing soil or rock.
- Excessive drawdown of groundwater increases the effective overburden on the foundation soils underneath the MSWLF unit, which may cause excessive settlement or bearing capacity failure on the foundation soils.
- A closed landfill as the foundation for a new landfill ("piggy-backing") may be unstable unless the closed landfill has undergone complete settlement of the underlying wastes.

2.7.1. *Site Stability Assessment*

For a landfill site in an unstable area, owners/operators must assess the ability of the soils and/or rock to serve as a foundation as well as the ability of the site embankments and slopes to maintain a stable condition. Once these factors have been evaluated, an MSWLF design should be developed that will address these types of concerns and prevent possible associated damage to MSWLF structural components. In designing a new unit or lateral expansion or re-evaluating an existing MSWLF unit, a stability assessment should be conducted in order to avoid or prevent a destabilizing event from impairing the structural integrity of the landfill component systems. A stability assessment involves essentially three components: (i) evaluation of subsurface conditions, (ii) analysis of slope stability, and (iii) examination of related design needs. Evaluation of subsurface conditions requires:

- Assessing the stability of foundation soils, adjacent embankments, and slopes;
- Investigating the geotechnical and geological characteristics of the site to establish soil strengths and other engineering properties by performing standard penetration tests, field vane shear tests, and laboratory tests; and
- Testing the soil properties such as water content, shear strength, plasticity, and grain size distribution.

Stability assessment should consider:

- The adequacy of the subsurface exploration program;
- The liquefaction potential of the embankment, slopes, and foundation soils;
- The expected behavior of the embankment, slopes, and foundation soils when they are subjected to seismic activity;
- The potential for seepage-induced failure; and
- The potential for differential settlement.

In addition, must assess at a minimum natural conditions (i.e. soil, geology, and geomorphology) as well as human-made features or events (both subsurface and surface) that could cause differential settlement of ground. Natural conditions can be highly unpredictable and destructive, especially if amplified by human-induced changes to the environment. Specific examples of natural or human-induced phenomena include: debris flows resulting from heavy rainfall in a small watershed; the rapid formation of a sinkhole because of excessive local or regional groundwater withdrawal in a limestone region; earth displacement by faulting activity; and rockfalls along a cliff face caused by vibrations resulting from the detonation of explosives or sonic booms. However, information on natural features can be obtained from the geographical map; regional or local soil maps; aerial photographs (especially in karst areas); GIS and satellite images; site-specific investigations; etc. To examine an area for possible sources of human-induced ground instability, the site and surrounding area should be examined for activities related to extensive withdrawal of oil, gas, or water from subsurface units as well as construction or other operations that may result in ground motion (i.e. blasting).

2.7.2. *Site Failures Nature*

Failures occur when the driving forces imposed on the soils or engineered structures exceed the resisting forces of the material. The ratio of the resisting force to the driving force is considered the FS. At an FS value of <1.0, failure will occur by definition. There is a high probability that, due to natural variability and the degree of accuracy in measurements, interpreted soil conditions will not be precisely representative of the actual soil conditions. Therefore, failure may not occur exactly at the calculated value, so factors of safety >1.0 are required for the design. For plastic soils such as clay, movement or deformation (i.e. creep) may occur at a higher FS prior to catastrophic failure. Principal modes of failure in soil or rock include:

- Rotation (change of orientation) of an earthen mass on a curved slip surface approximated by a circular arc;
- Translation (change of position) of an earthen mass on a planar surface whose length is large compared to depth below ground;
- Displacement of a wedge-shaped mass along one or more planes of weakness;
- Earth and mud flows in loose clayey and silty soils; and
- Debris flows in coarse-grained soils.

Normally, three types of failures can occur at an MSWLF unit; they are settlement, loss of bearing strength, and sinkhole collapse:

- *Settlement* — If not properly engineered, a landfill in an unstable area may undergo extreme settlement, which can result in structural failure. Differential settlement is a particular mode of failure that generally occurs beneath a landfill in response to consolidation and dewatering of the foundation soils during and following waste loading. Settlement beneath a landfill unit, both total and differential, should be assessed and compared to the elongation strength and flexure properties of the liner and leachate collection pipe system. Even small amounts of settlement can seriously damage leachate collection piping and sumps. The analysis will provide an estimate of maximum settlement, which can be used to aid in estimating differential settlement. Allowable settlement is typically expressed as a function of total settlement because differential settlement is more difficult to predict. However, differential settlement is a more serious threat to the integrity of the structure than total settlement.
- *Loss of bearing strength* — It is a failure mode that tends to occur in areas that have soils that tend to expand, rapidly settle, or liquefy, thereby causing failure or reducing performance of overlying MSWLF components. Another example of loss of bearing strength involves failures that have occurred at operating sites where excavations for landfill expansions adjacent to the filled areas reduced the mass of the soil at the toe of the slope, thereby reducing the overall strength (resisting force) of the foundation soil.
- *Catastrophic collapse in the form of sinkholes* — It is a type of failure that occurs in karst regions. As water, especially acidic water percolates through limestone

> (calcium carbonate), the soluble carbonate material dissolves, forming cavities and caverns. Land overlying caverns can collapse suddenly, resulting in sinkhole features that can be 100 feet or more in depth and 300 feet or more in width.

Tables 5 and 6[19] provide some examples of analytical considerations for mode of failure assessments in both natural and human-made slopes.

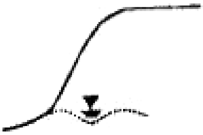
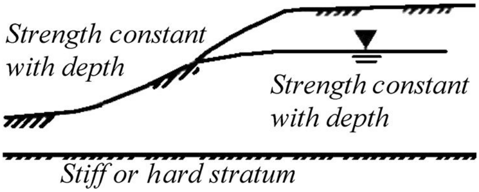
2.7.3. *Subsurface Exploration Programs*

Foundation soil stability assessments for noncatastrophic failure require field investigations to determine soil strengths and other soil properties. *In situ* field vane shear tests commonly are conducted in addition to collection of piston samples for laboratory testing of undrained shear strengths (biaxial and triaxial). Field vanes taken at depth provide a profile of soil strength. The required field vane depth intervals vary, based on soil strength and type, and the number of borings required depends on the variability of the soils, the site size, and landfill unit dimensions. Borings and field vane testing should consider the anticipated design to identify segments of the facility where critical cross sections are likely to occur. Critical sections are where factors of safety are anticipated to be the lowest. Other tests that are conducted to characterize a soil include determination of water content, *Atterberg limits*, grain-size distribution, consolidation, effective porosity, and saturated hydraulic conductivity. The site hydrogeologic conditions should be assessed to determine if soils are saturated or unsaturated. Catastrophic failures, such as sinkhole collapse in karst terrains or fault displacement during an earthquake, are more difficult to predict. Subsurface karst structures may have surface topographic expressions such as circular depressions over subsiding solution caverns. Subsurface borings or geophysical techniques may provide reliable means of identifying the occurrence, depth, and size of solution cavities that have the potential for catastrophic collapse.

2.7.4. *Slope Stability Analysis*

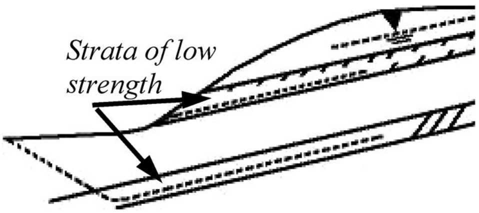
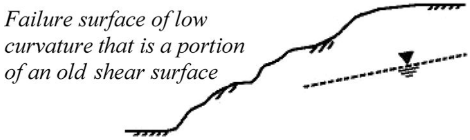
Slope stability analyses are performed for both excavated side slopes and above ground embankments. The analyses are performed as appropriate to verify the structural integrity of a cut slope or dike. The design configuration is evaluated for its stability under all potential hydraulic and loading conditions, including conditions that may exist during construction of an expansion (i.e. excavation). Analyses typically performed are slope stability, settlement, and liquefaction. FS rationale and selection for different conditions are described by Huang[21] and Terzaghi and Peck[22] as well as numerous publications in the field of soil mechanics and geotechnical research areas. Lists in Table 7[19] recommended minimum FS values for slopes. However, FS may vary with regional geological or geomorphological formation; therefore,

Table 5. Analysis of Stability of Some Natural Slopes.[19]

1. Slope in coarse-grained soil with some cohesion *Low Grundwater* Failure of thin wedge, position influenced by tension cracks *High Groundwater* Failure at relatively shallow toe circles  	• With low groundwater, failure occurs on shallow, straight, or slightly curved surface. The presence of a tension crack at the top of the slope influences failure location. With high groundwater, failure occurs on the relatively shallow toe circle whose position is determined primarily by ground elevation. • Analyze with effective stress using strengths C' and $\text{Ø}'$ from CD tests. Pore pressure is governed by seepage condition. Internal pore pressures and external water pressures must be included
2. Slope in coarse-grained, soil cohesion *Low Grundwater* Stable slope angle = effective friction angle *High Groundwater* Stable slope angle = $\frac{1}{2}$ effective friction angle  	• Stability depends primarily on groundwater conditions. With low groundwater, failures occur as surface sloughing until slope angle flattens to friction angle. With high groundwater, stable slope is approximately $\frac{1}{2}$ friction angle. • Analyze with effective stress using strengths C' and $\text{Ø}'$ from CD tests. Slight cohesion appearing in test envelope is ignored. Special consideration must be given to possible flow slides in loose and saturated fine sands.
3. Slope in normally consolidated or slightly pre-consolidated clay *Location of failure depends on variation of shear strength with depth.*  	• Failure occurs on circular arcs whose position is governed by theory. The position of groundwater table does not influence stability unless its fluctuation changes the strength of the clay or acts in tension cracks. • Analyze with total stresses, zoning cross section for different values of shear strengths. Determine shear strength from unconfined compression test, unconsolidated undrained triaxial test, or vane shear.

(*Continued*)

Table 5. (*Continued*)

4. Slope in stratified soil profile *Location of failure depends on relative strength and orientation of layers.* 	• The location of failure plane is controlled by relative strength and orientation of strata. Failure surface is the combination of active and passive wedges with central sliding block chosen to conform to stratification. • Analyze with effective stress using strengths C′ and Ø′ for fine-grained strata and Ø′ for cohesionless material.
5. Depth creep movements in old slide mass *Bowl-shaped area of low slope (9–11%) bounded at top by old scarp.* 	• Strength of old slide mass decreases with the magnitude of movement that has occurred previously. Most dangerous situation is in stiff and over-consolidated clay that is softened, fractured, or slickensided in the failure zone.

it will need to modify with local adjustment and provide their own minimum FS requirements.

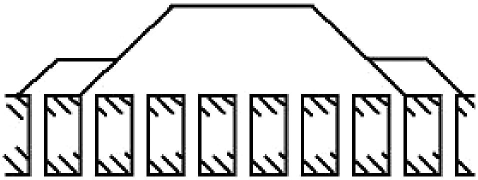
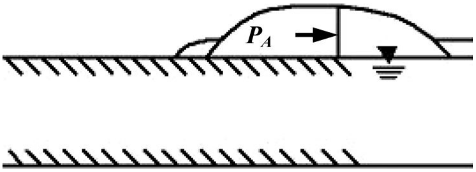
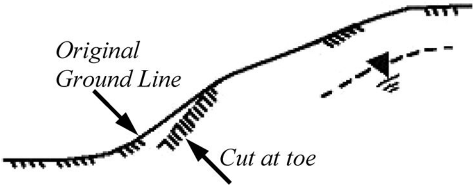
A computer program that is widely used for slope stability analysis is PC STABL, which is a two-dimensional (2D) model that computes the minimum critical factors of safety between layer interfaces. This model uses the method of vertical slices to analyze the slope and calculate the FS. The PC STABL can account for heterogeneous soil systems, anisotropic soil strength properties, excess pore water pressure due to shear, static groundwater and surface water, pseudostatic earthquake loading, surcharge boundary loading, and tieback loading.

2.7.5. *Design Considerations for Slope Stabilization*

Some methods for slope stabilization are presented in Table 8[19] and are summarized below.

- The first illustration in Table 8 shows that stability can be increased by changing the slope geometry through reduction of the slope height, flattening the slope angle, or excavating a bench in the upper part of the slope.
- The second illustration in Table 8 shows how compacted earth or rock fill can be placed in the form of a berm at and beyond the slope's toe to buttress the slope. To prevent the development of undesirable water pressure behind the

Table 6. Analysis of Stability for Cut and Fill Slopes (Condition Varying with Time).[19]

1. Failure of fill on soft cohesive foundation with sand drains  *Location of failure depends on geometry and strength of cross section.*	• Usually, minimum stability occurs during placing of fill. If the rate of construction is controlled, allow for gain in strength with consolidation from drainage. • Analyze with effective stress using strengths C′ and Ø′ from CU tests with pore pressure measurement. Apply estimated pore pressures or piezometric pressures. Analyze with total stress for rapid construction without the observation of pore pressures, use shear strength from unconfined compression or unconsolidated undrained triaxial.
2. Failure of stiff compacted fill on soft cohesive foundation  *Failure surface may be rotation on circular arc or translation with active and passive wedges.*	• Usually, minimum stability obtained at the end of construction. Failure may be in the form of rotation or translation, and both should be considered. • For rapid construction, ignore consolidation from drainage and utilize shear strengths determined from U or UU tests or vane shear in total stress analysis. If failure strain of fill and foundation materials differ greatly, safety factor should exceed one, ignoring shear strength of fill. Analyze long-term stability using C and Ø from CU tests with effective stress analysis, applying the pore pressures of groundwater only.
3. Failure following cut in stiff fissured clay  *Failure surface depends on pattern of fissures or depth of softening.*	• The release of horizontal stresses by excavation causes the expansion of clay and opening of fissures, resulting in loss of cohesive strength. • Analyze for short-term stability using C′ and Ø′ with total stress analysis. Analyze for long-term stability with C'_r and $Ø'_m$ based on residual strength measured in consolidated drained tests.

berm, a drainage system may be placed behind the berm at the base of the slope.

- The third illustration in Table 8 presents several types of retaining structures. These structures generally involve drilling and/or excavation followed by constructing cast-in-place concrete piles and/or slabs.

Table 7. Recommended Minimum Limit of Factor of Safety for Slope Stability Analyses.[19]

	Uncertainty of Strength Measurements	
Consequences of Slope Failure	Small[a]	Large[b]
No imminent danger to human life or major environmental impact if slope fails	1.25 (1.20)[c]	1.50 (1.30)
Imminent danger to human life or major environmental impact if slope fails	1.50 (1.30)	2.00 or greater (1.70 or greater)

[a] The uncertainty of the strength measurements is smallest when the soil conditions are uniform and high-quality strength test data provide a consistent, complete, and logical picture of the strength characteristics.

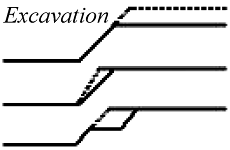
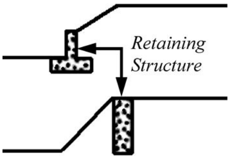
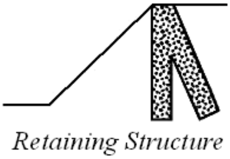
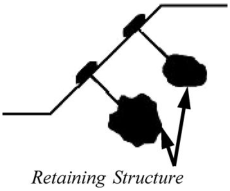
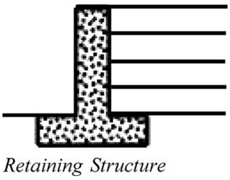
[b] The uncertainty of the strength measurements is greatest when the soil conditions are complex and when available strength test data do not provide a consistent, complete, and logical picture of the strength characteristics.

[c] Number without parentheses apply for static conditions and those within parentheses apply to seismic conditions.

- The T-shaped cantilever wall design enables some of the retained soil to contribute to the stability of the structure and is advisable for use on slopes that have vertical cuts.
- Closely spaced vertical piles placed along the top of the slope area provide reinforcement against slope failure through a soil arching effect that is created between the piles. This type of retaining system is advisable for use on steeply cut slopes.
- Vertical piles also may be designed with a tie back component at an angle to the vertical to develop a high resistance to lateral forces. This type of wall is recommended for use in areas with steeply cut slopes where soil arching can be developed between the piles.
- The last retaining wall shown uses a cantilever setup along with soil that has been reinforced with geosynthetic material to provide a system that is highly resistant to vertical and lateral motion. This type of system is best suited for use in situations where vertically cut slopes must have lateral movement strictly controlled.

Other potential procedures for stabilizing natural and human-made slopes include the use of geotextiles and geogrids to provide additional strength; the installation of wick and toe drains to relieve excess pore pressures, grouting, and vacuum and well-point pumping to lower groundwater levels. In addition, surface drainage may be controlled to decrease infiltration, thereby reducing the potential for mud and debris slides in some areas. Lowering the groundwater table also may have stabilizing effects. Walls or large-diameter piling can be used to stabilize slides of

Table 8. Some Methods of Stabilizing Excavation Slopes.[19]

Scheme	Applicable Methods	Comments
1. Changing geometry 	1. Reduce slope height by excavation at top of slope 2. Flatten the slope angle. 3. Excavate a bench in upper part of slope.	• Area has to be accessible to construction equipment. Disposal site needed for excavated soil. Drainage sometimes incorporated in this method.
2. Earth berm fill 	1. Compacted earth or rock berm placed at end beyond the toe. Drainage may be provided behind the berm.	Sufficient width and thickness of berm required so failure will not occur below or through the berm.
3. Retaining structures 	1. Retaining wall: crib or cantilever type. 2. Drilled and cast-in-place vertical piles and/or slabs founded well below bottom slide plane. Generally, 18–36 in. in diameter and 4–8 ft spacing. Larger diameter piles at closer spacing may be required in some cases with mitigate failures of cuts in highly fissured clays.	• Usually expensive. Cantilever walls might have to be tied back. • Spacing should be such that soil can arch between piles. Grade beam can be used to tie piles together. Very large diameter (6 ft ±) piles have been used for deep slide.
	3. Drilled and cast-in-place vertical piles tied back with battered piles or a deadman. Piles founded well below slide plane. Generally, 12–30 in. in diameter and at least 4- to 8-foot spacing.	• Space close enough so soil will arch between piles. Piles can be tied together with grade beam.
	4. Earth and rock anchors and rock bolts.	• Can be used for high slopes, and in very restricted areas. Conservative design should be used, especially for permanent support. Use may be essential for slopes in rocks where joints dip toward excavation, and such joints daylight in the slope.
	5. Reinforced earth.	• Usually expensive.

relatively small dimension or to retain steep toe slopes, so that failure will not extend back into a larger mass.

During construction activities, it may be appropriate to monitor slope stability because of the additional stresses placed on natural and engineered soil systems (e.g., slopes, foundations, and dikes) as a result of excavation and filling activities. Post-closure slope monitoring usually is not necessary. Important monitoring parameters may include settlement, lateral movement, and pore water pressure. Monitoring for pore water pressure is usually accomplished with piezometers screened in the sensitive strata. Lateral movements of structures may be detected on the surface by surveying horizontal and vertical movements. Subsurface movements may be detected by the use of slope inclinometers. Settlement may be monitored by surveying ground surface elevations (on several occasions over a period of time) and comparing them with areas that are not likely to experience changes in elevations.

3. Operating Crieria

3.1. *General Overview*

The SWDFC contain a series of operating requirements pertaining to routine operation, management, and environmental monitoring at MSWLF units. The operating requirements pertain to new MSWLF units, existing MSWLF units, and lateral expansions of existing MSWLF units. The operating requirements have been developed to ensure the safe daily operation and management at MSWLF units. The operating requirements include:

- Exclusion of hazardous waste;
- Cover material;
- Disease vector control;
- Explosive gases control;
- Air monitoring;
- Facility access;
- Run-on/run-off control systems;
- Surface water requirements;
- Liquid restrictions; and
- Recordkeeping requirements.

3.2. *Exclusion of Hazardous Waste*

The MSWLF hazardous waste exclusion program should be capable of detecting and preventing disposal of regulated hazardous wastes and polychlorinated biphenyls (PCB) wastes that may be liquids or nonliquids (sludges or solids). PCB wastes do not include small capacitors found in fluorescent light ballast, white goods (i.e. washers, dryers, refrigerators), or other consumer electrical products (e.g., radio and television units).

The owner or operator must develop a program to detect and prevent disposal of regulated hazardous wastes or PCB wastes at the MSWLF facility. Hazardous wastes may be gases, liquids, solids, or sludges as well characteristics of hazardous wastes that include ignitability, corrosivity, reactivity, and toxicity. Household hazardous wastes are excluded and the wastes generated by conditionally exempt small quantity generators (CESQGs) are not considered as regulated hazardous wastes; therefore, these wastes may be accepted for disposal at an MSWLF unit. The hazardous waste exclusion program is not intended to identify whether regulated hazardous waste or PCB waste was received at the MSWLF unit. Commercial or industrial sources of PCB wastes that should be addressed by the program include:

- Mineral oil and dielectric fluids containing PCBs;
- Contaminated soil, dredged material, sewage sludge, rags, and other debris from a release of PCBs;
- Transformers and other electrical equipment containing dielectric fluids; and
- Hydraulic machines.

The owner or operator is required to implement a program to detect and exclude regulated hazardous wastes and PCBs from disposal in the MSWLF unit(s). This program must include elements for:

- Random inspections of incoming loads or other prevention methods;
- Maintenance of inspection records;
- Facility personnel training; and
- Notification to appropriate authorities if hazardous wastes or PCB wastes are detected.
- Each of these program elements is discussed separately on the following pages.

Ideally, all loads should be screened; however, it is generally not practical to inspect in detail all incoming loads. An inspection is typically a visual observation of the incoming waste loads by an individual who is trained to identify regulated hazardous or PCB wastes that would not be acceptable for disposal at the MSWLF unit. An inspection flowchart to identify, accept, or refuse solid waste is provided as Fig. 9.[23]

The waste should be carefully spread for observation using a front-end loader or other piece of equipment. Personnel should be trained to identify suspicious wastes. Some indications of suspicious wastes are:

- Hazardous placards or marking;
- Liquids;

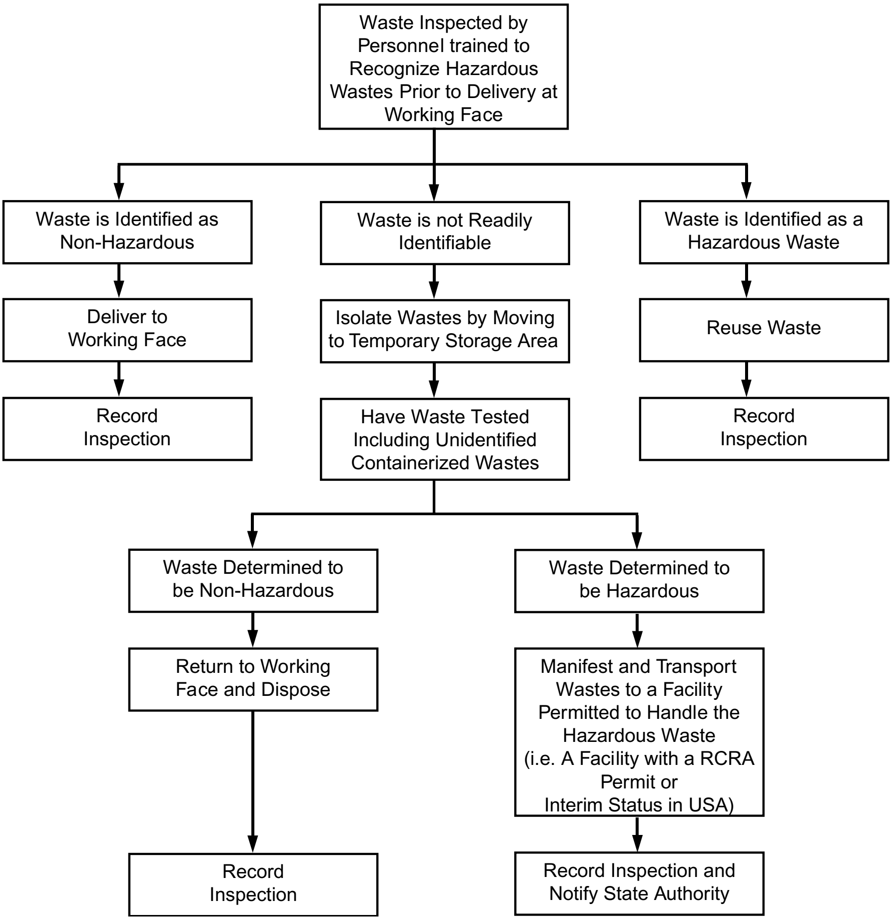


Figure 9. Hazardous Waste Inspection Decision Tree (Inspection Prior to Working Face).[23]

- Powders or dusts;
- Sludges;
- Bright or unusual colors;
- Drums or commercial size containers; or
- Chemical odors.

The owner or operator should develop specific procedures to be followed when suspicious wastes are discovered. The procedure should include the following points:

- Segregate the wastes;
- Question the driver;

- Review the manifest (if applicable);
- Contact possible source;
- Call the appropriate state or federal agencies;
- Use appropriate protective equipment;
- Contact laboratory support if required; and
- Notify a response agency if necessary.

Hazardous waste may be stored at the MSWLF facility for 90 days to fulfill the following requirements, which are:

- The waste is placed in tanks or containers;
- The date of receipt of the waste is clearly marked and visible on each container;
- The container or tank is marked clearly with the words “Hazardous Waste”;
- An employee is designated as the emergency coordinator who is responsible for coordinating all emergency response measures; and
- The name and telephone number of the emergency coordinator and the number of the fire department is posted next to the facility phone.

PCB wastes detected at an MSWLF facility must be stored and disposed. Therefore, the owner or operator is required to:

- Obtain an EPA PCB identification number;
- Properly store the PCB waste;
- Mark containers or items with the words “Caution: Contains PCBs”; and
- Manifest the PCB waste for shipment to a permitted incinerator, chemical waste landfill, or high-efficiency boiler (depending on the nature of the PCB waste) for disposal.

3.3. *Cover Material Requirement*

There are three types of covers those are used in an MSWLF:

1. *Daily cover* — This is a layer of soil that is placed on top of the landfill at the end of each day.
2. *Intermediate cover* — This cover is used on the top areas of a landfill that will not be used for a long period (about two years or more). Covering the parts of a landfill that are not used will reduce the amount of liquid that will accumulate in the landfill, since any water that accumulates on the surface of an intermediate cover *is* allowed to run off the site and onto the surrounding land. Intermediate covers are usually made from any type of soil that is available at the landfill site. The thickness of the intermediate cover is approximately 12″, twice as much as is used for the daily cover. Vegetation is usually allowed to grow on top of the

cover as protection against erosion. When the area is to be used again for dumping waste, the vegetation is scraped off.

3. *Final cover* — The final cover on a landfill is meant to seal the landfill and to reduce the amount of water that will enter the landfill after it is closed. It usually consists of five layers of material.

Owners and operators of new MSWLF units, existing MSWLF units, and lateral expansions are required to cover solid waste at the end of each operating day with six inches of earthen material. This cover material requirement is not related to the final cover required. The placement of six inches of cover controls disease vectors (i.e. birds, insects, or rodents that represent the principal transmission pathway of a human disease) by preventing egress from the waste and by preventing access to breeding environments or food sources. Covering also reduces exposure of combustible materials to ignition sources and may reduce the spread of fire if the disposed waste burns. Odors and blowing litter are reduced by eliminating the direct contact of wind and disposed waste. Similarly, scavenging is reduced by removing the waste from observation. Should these unwanted effects of inadequate cover persist, the owner or operator may increase the amount of soil used or apply it more frequently. Any soil type can meet the requirements of the regulation when placed in a six-inch layer.

Some alternative materials are also allowed for daily cover. There are no numerical requirements for the alternative cover; rather, the alternative cover must control disease vectors, fires, odors, blowing litter, and scavenging without presenting a threat to human health and the environment. Some suggested methods for demonstrating alternative covers are: (i) side-by-side (six inches of earthen materials and alternative cover) test pads; (ii) full-scale demonstration; and (iii) short-term full-scale tests. Alternative daily cover materials may include indigenous materials or commercially available materials. Indigenous materials are those materials that would be disposed as waste; therefore, using these materials is an efficient use of landfill space. Examples of indigenous materials include:

- Ash from municipal waste combustors and utility companies;
- Compost-based material;
- Foundry sand from the manufacturing process of discarding used dies;
- Yard waste such as lawn clippings, leaves, and tree branches;
- Sludge-based materials (i.e., sludge treated with lime and mixed with ash or soil);
- Construction and demolition debris (which has been processed to form a slurry);
- Shredded automobile tires;
- Discarded carpets; and
- Grit from municipal wastewater treatment plants.

Some of the types of commercially available daily cover materials include:

- Foam that usually is sprayed on the working face at the end of the day;
- Geosynthetic products such as a tarp or fabric panel that is applied at the end of the working day and removed at the beginning of the following working day; and
- Slurry products (e.g., fibers from recycled newspaper and wood chip slurry, clay slurry).

Other criteria to consider when selecting an alternative daily cover material include availability and suitability of the material, equipment requirements, and cost. The temporary climatic waiver of the cover requirement is available only to owners or operators in some specific climatic conditions. Activities that may be affected by extreme seasonal climatic conditions include:

- Obtaining cover soil from a borrow pit;
- Transporting cover soil to the working face; or
- Spreading and compacting the soil to achieve the required functions.

Extremely cold conditions may prevent the efficient excavation of soil from a borrow pit or the spreading and compaction of the soil on the waste. Extremely wet conditions (e.g., prolonged rainfall, flooding) may prevent transporting cover soil to the working face and may make it impractical to excavate or spread and compact. The duration of waivers may be as short as one day for unusual rain storms, or as long as several months for extreme seasonal climatic conditions.

3.4. *Disease Vector Control*

Disease vectors such as rodents, birds, flies, and mosquitoes typically are attracted by putrescent waste and standing water, which act as food source and breeding ground. Putrescent waste is solid waste that contains organic matter (such as food waste) capable of being decomposed by microorganisms. An MSWLF facility typically accepts putrescent wastes. Application of cover at the end of each operating day generally is sufficient to control disease vectors; however, other vector control alternatives may be required. These alternatives could include: reducing the size of the working face; other operational modifications (i.e., increasing cover thickness, changing cover type, density, placement frequency, and grading); repellents, insecticides, or rodenticides; composting or processing of organic wastes prior to disposal; and predatory or reproductive control of insect, bird, and animal populations.

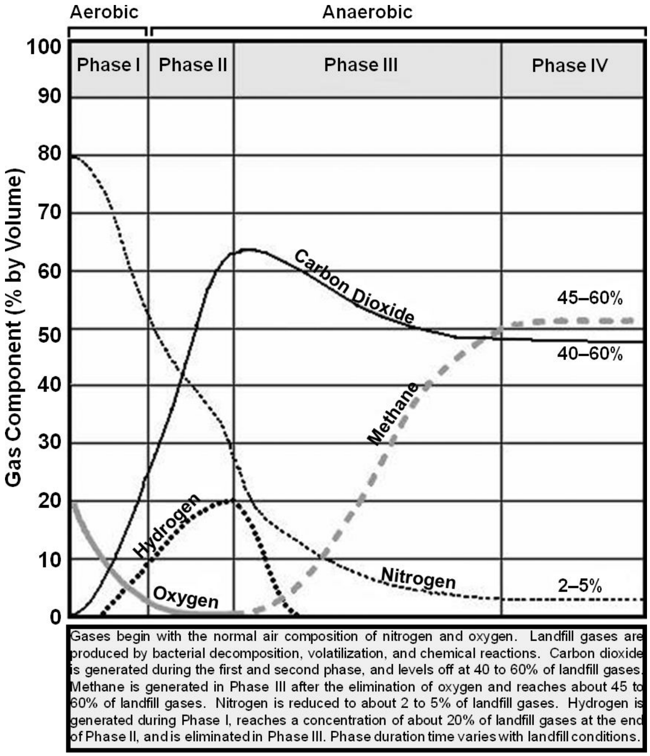
Mosquitoes, for example, are attracted by standing water found at MSWLFs, which can provide a potential breeding ground after only three days. Water generally collects in surface depressions, open containers, exposed tires, ponds resulting from soil excavation, leachate storage ponds, and siltation basins. Landfill operations that minimize standing water and that use an insecticide spraying program ordinarily are effective in controlling mosquitoes. Vectors may reach the landfill facility not only from areas adjacent to the landfill, but through other modes conducive to harborage and breeding of disease vectors. Such modes may include residential and commercial route collection vehicles and transfer stations. These transport modes and areas also should be included in the disease vector control program if disease vectors at the landfill facility become a problem. Keeping the collection vehicles and transfer stations covered; emptying and cleaning the collection vehicles and transfer stations; using repellents, insecticides, or rodenticides; and reproductive control are all measures available to reduce disease vectors in these areas.

3.5. *Explosive Gases Control*

Landfill gases are the result of microbial decomposition of solid waste. Gases produced include methane (CH_4), carbon dioxide (CO_2), and lesser amounts of other gases [i.e. hydrogen (H), volatile organic compounds (VOCs), and hydrogen sulfide (H_2S)].[24] Methane gas, the principal component of natural gas, is generally the primary concern in evaluating landfill gas generation, as it is odorless and highly combustible. Typically, H gas is present at much lower concentrations. H gas forms as decomposition progresses from the acid production phase (i.e. aerobic condition) to the methanogenic phase (i.e. anaerobic condition). While H is explosive and is occasionally detected in landfill gas, it readily reacts to form CH_4 or H_2S. The H_2S is toxic and is readily identified by its "rotten egg" smell at a threshold concentration near 5 ppb (parts per billion). Figure 10[25] illustrates MSWLFs gas generation and emission illustrate in different phase conditions.

The accumulation of CH_4 in MSWLF structures can potentially result in fire and explosions that can endanger employees, users of the disposal site, and occupants of nearby structures, or cause damage to landfill containment structures. These hazards are preventable through monitoring and through corrective action should CH_4 gas levels exceed specified limits in the facility structures (i.e. excluding gas control or recovery system components), or at the facility property boundary. Therefore, MSWLF facility owners and operators must comply with the following requirements:

- Monitor at least quarterly;
- Take immediate steps to protect human health in the event of CH_4 gas levels exceeding 25% of the lower explosive limit (LEL) in facility structures, such as evacuating the building;
- Notify the concern authorities if CH_4 levels exceed 25% of the LEL in facility structures or exceed the LEL at the facility property boundary;



Gases begin with the normal air composition of nitrogen and oxygen. Landfill gases are produced by bacterial decomposition, volatilization, and chemical reactions. Carbon dioxide is generated during the first and second phase, and levels off at 40 to 60% of landfill gases. Methane is generated in Phase III after the elimination of oxygen and reaches about 45 to 60% of landfill gases. Nitrogen is reduced to about 2 to 5% of landfill gases. Hydrogen is generated during Phase I, reaches a concentration of about 20% of landfill gases at the end of Phase II, and is eliminated in Phase III. Phase duration time varies with landfill conditions.

Figure 10. Gas Generation and Emissions in Different Landfill Phase Conditions.[25]

- Within seven days of detection, place in the operating record documentation that CH_4 gas concentrations exceeded the criteria, along with a description of immediate actions taken to protect human health; and
- Within 60 days of detection, implement a remediation plan for the CH_4 gas releases, notify the concern authorities, and place a copy of the remediation plan in the operating record.

Landfill gas production rates vary spatially within a landfill unit because of pockets of elevated microbial activity but, due to partial pressure gradients, differences in gas composition are reduced as the gases commingle within and outside the landfill unit. Although CH_4 gas is lighter than air and CO_2 is heavier, these gases are concurrently produced at the microbial level and will not separate by their individual density. These gases will remain mixed and migrate according to the density gradients between the landfill gas and the surrounding gases (i.e., a mixture of CH_4 and CO_2 in a landfill unit or in surrounding soil will not separate by rising and sinking respectively, but will migrate as a mass in accordance with the density of the mixture and other gradients such as temperature and partial pressure). When undergoing vigorous microbial production, gas pressures on the order of 1–3 inches of water [a barometric pressure change of 2″ of mercury is equivalent to 27.2″ of water] relative to atmospheric pressure are common at landfill facilities, with much higher pressures occasionally reported. Negative gas pressures are commonly observed and are believed to occur because of the delayed response within a landfill unit to the passage of a high-pressure system outside the landfill unit. Barometric highs will tend to introduce atmospheric oxygen into surface soils in shallow portions of the landfill unit, which may alter microbial activity, particularly methane production and gas composition. Migration of landfill gas is caused by concentration gradients, pressure gradients, and density gradients. The direction in which landfill gas will migrate is controlled by the driving gradients and gas permeability of the porous material through which it is migrating. Figure 11[23] illustrates the potential effects of surrounding geology on gas migration.

Generally, landfill gas will migrate through the path of least resistance. Coarse and porous soils such as sand and gravel will allow greater lateral migration or transport of gases than finer-grained soils. Moreover, resistance to landfill gas flow increases, as moisture content increases and, therefore, an effective barrier to gas flow can be created under saturated conditions. Thus, readily drained soil conditions, such as sands and gravels above the water table, may provide a preferred flow-path, but unless finer-grained soils are fully saturated, landfill gases also can migrate in a "semi-saturated" zone. While geomembranes (i.e. synthetic) may not eliminate

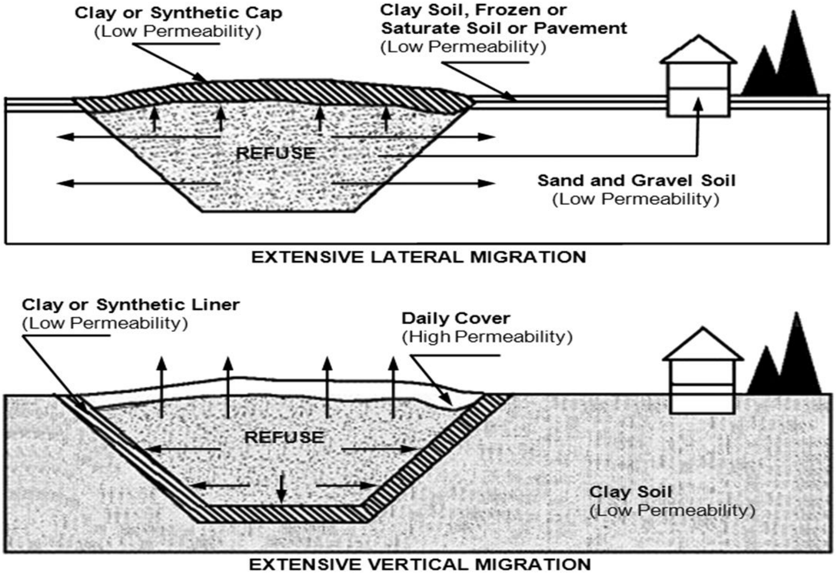


Figure 11. Potential Effects of Surrounding Geology on Gas Migration[23]: (a) Extensive Lateral Migration and (b) Extensive Vertical Migration.

landfill gas migration, landfill gas in a closed MSWLF unit will tend to migrate laterally if the final cover contains a geomembrane (i.e. synthetic) and if the side slopes of the landfill do not contain an effective gas barrier. Lateral gas migration is more common in older facilities that lack appropriate gas control systems. The degree of lateral migration in older facilities also may depend on the type of natural soils surrounding the facility. Stressed vegetation may indicate gas migration. Landfill gas present in the soil atmosphere tends to make the soil anaerobic by displacing the oxygen, thereby asphyxiating the roots of plants. The higher the concentration of combustible gas and/or CO_2 and the lower the amount of oxygen are, the greater will be the extent of damage to vegetation.

3.5.1. *Gas Monitoring*

The owner or operator of an MSWLF unit/facility must implement routine CH_4 gas monitoring program to comply with the LEL requirements for CH_4. The CH_4 is explosive when present in the range of 5–15% by volume in air. When present in air at concentrations greater than 15%, the mixture will not explode. This 15% threshold value is the upper explosive limit (UEL). The UEL is the maximum concentration of a gas or vapor above which the substance will not explode when exposed to a source of ignition. The explosive hazard range is between the LEL and the UEL. However, CH_4 concentrations above the UEL remain a significant concern; fire and asphyxiation can still occur at these levels. In addition, even a minor dilution of the methane by increased ventilation can bring the mixture back into the explosive range. To demonstrate compliance, the owner/operator would sample air within facility structures where gas may accumulate and in soil at the property boundary. Other monitoring methods may include: (1) sampling gases from probes within the landfill unit or within the LCS or (2) sampling gases from monitoring probes installed in soil between the landfill unit and either the property boundary or structures where gas migration may pose a danger. A typical gas-monitoring probe installation is depicted in Fig. 12.[23]

Although collection of data, such as water presence and level, gas probe pressure, ambient temperature, barometric pressure, and the occurrence of precipitation during sampling, provides useful information in assessing monitoring results. For example, falling barometric pressure may cause increased subsurface (gas) pressures and corresponding increased methane content as gas more readily migrates from the landfill. Gas probe pressure can be measured using a portable gauge capable of measuring both vacuum and pressure in the range of 0–5 inches of water pressure (or other suitable ranges for pressure conditions); this pressure should be measured prior to methane measurement or sample collection in the gas probe. A representative sample of formation (subsurface) gases can be collected directly from the probe.

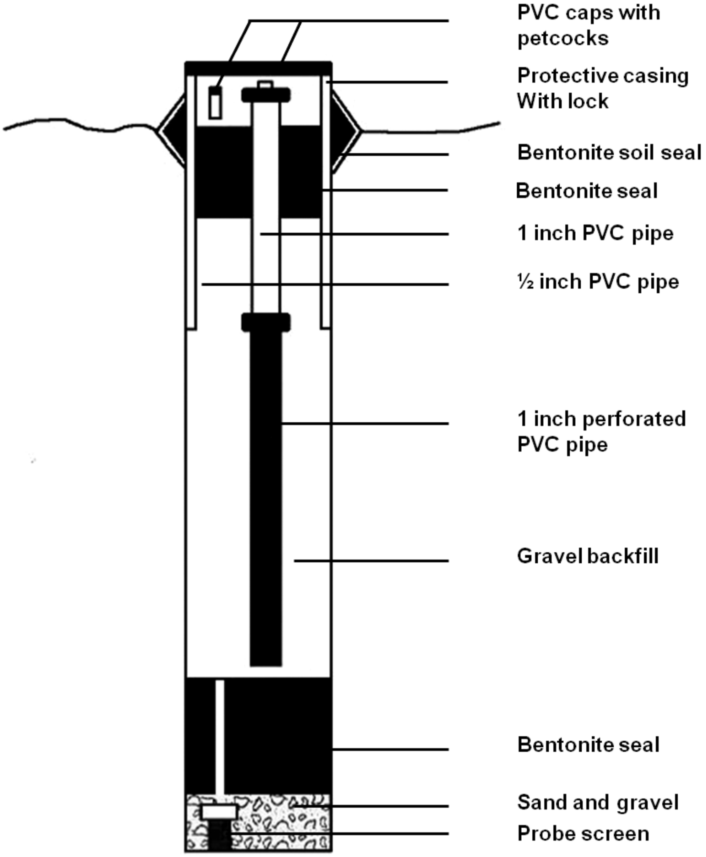


Figure 12. Typical Gas Monitoring Probe.[23]

Purging typically is not necessary due to the small volume of the probe. A water trap is recommended to protect instrumentation that is connected directly to the gas probe. After measurements are obtained, the gas probe should be capped to reduce the effects of venting or barometric pressure variations on gas composition in the vicinity of the probe.

The frequency of monitoring should be sufficient to detect landfill gas migration based on subsurface conditions and changing landfill conditions such as partial or complete capping, landfill expansion, gas migration control system operation or failure, construction of new or replacement structures, and changes in landscaping or land use practices. The rate of landfill gas migration because of these anticipated changes and the site-specific condition provides the basis for establishing monitoring frequency. Monitoring is to be conducted at least quarterly. The number and location of gas probes is also site-specific and highly dependent on subsurface conditions, land use, and location and design of facility structures. Monitoring for gas migration should be within the more permeable strata. Multiple or nested probes are useful in defining the vertical configuration of the migration pathway. Structures with basements or crawl spaces are more susceptible to landfill gas infiltration. Elevated structures are typically not at risk.

Measurements are usually made in the field with a portable methane meter, explosimeter, or organic vapor analyzer. Gas samples also may be collected in glass or metal containers for laboratory analysis. Instruments with scales of measure in "percentage of LEL" can be calibrated and used to detect the presence of methane. Instruments of the hot-wire Wheatstone bridge type (i.e., catalytic combustion) directly measure combustibility of the gas mixture withdrawn from the probe. The thermal conductivity type meter is susceptible to interference as the relative gas composition and, therefore, the thermal conductivity, changes. Field instruments should be calibrated prior to measurements and should be rechecked after each day's monitoring activity.

Laboratory measurements with organic vapor analyzers or gas chromatographs may be used to confirm the identity and concentrations of gas. In addition to measuring gas composition, other indications of gas migration may be observed. These include odor (generally described as either a "sweet" or a rotten egg (H_2S) odor), vegetation damage, septic soil, and audible or visual venting of gases, especially in standing water. Exposure to some gases can cause headaches and nausea. If methane concentrations are in excess of 25% of the LEL in facility structures or exceed the LEL at the property boundary, the danger of explosion is imminent. Immediate action must be taken to protect human health from potentially explosive conditions. All personnel should be evacuated from the area immediately. Venting the building upon exit (e.g., leaving the door open) is desirable but should not replace evacuation procedures.

As per regulations of US EPA, owners and operators in unapproved states have 60 days after exceeding the CH_4 level to prepare and implement a remediation plan. The remediation plan should describe the nature and extent of the CH_4 problem as well as a proposed remedy. To comply with this 60-day schedule, an investigation of subsurface conditions may be needed in the vicinity of the monitoring probe where the criterion was exceeded. The objectives of this investigation should be to describe the frequency and lateral and vertical extents of excessive CH_4 migration (which exceeds the criterion). Such an investigation also may yield additional characterization of unsaturated soil within the area of concern. The investigation should consider possible causes of the increase in gas concentrations such as landfill operational procedures, gas control system failure or upset, climatic conditions, or closure activity. Based on the extent and nature of the excessive CH_4 migration, a remedial action should be described, if the exceedance is persistent, that can be implemented within the prescribed schedule. The 60-day schedule does not address the protection of human health and the environment. The owner or operator still must take all steps necessary to ensure protection of human health, including interim measures.

3.5.2. *Landfill Gas Control System*

Landfill gas may vent naturally or be purposely vented to the atmosphere by vertical and/or lateral migration controls. Systems used to control or prevent gas migration are categorized as either passive or active systems.

3.5.2.1. Passive systems

Passive gas control systems rely on natural pressure and convection mechanisms to vent landfill gas to the atmosphere. Passive systems typically use "high-permeability" or "low-permeability" techniques, either singularly or in combination for a site. High-permeability systems use conduits such as ditches, trenches, vent wells, or perforated vent pipes surrounded by coarse soil to vent landfill gas to the surface and the atmosphere. Low-permeability systems block lateral migration through barriers such as synthetic membranes and high moisture-containing fine-grained soils. Passive systems may be incorporated into a landfill design or may be used for remedial or corrective purposes at both closed and active landfills. They may be installed within a landfill unit along the perimeter, or between the landfill and the disposal facility property boundary. A passive system may be incorporated into the final cover system of a landfill closure design and may consist of perforated gas collection pipes, high-permeability soils, or high-transmissivity geosynthetics located just below the low-permeability gas and hydraulic barrier or infiltration layer in the cover system. These systems may be connected to vent pipes that vent gas through the cover system or that are connected to header pipes located along the perimeter of the landfill unit. Figure 13[23] illustrates a passive system. The landfill gas collection system also may be connected with the LCS to vent gases in the headspace of leachate collection pipes.

Some problems have been associated with passive systems. For example, snow and dirt may accumulate in vent pipes, which will be preventing gas from venting. Vent pipes at the surface are susceptible to clogging by vandalism. Biological clogging of the system is also more common in passive systems.

3.5.2.2. Active systems

Active gas control systems use mechanical means to remove landfill gas and consist of either positive pressure (air injection) or negative pressure (extraction) systems. Positive pressure systems induce a pressure greater than the pressure of the migrating gas and drive the gas out of the soil and/or back to the landfill unit in a controlled manner. Negative pressure systems extract gas from a landfill by using a blower to pull gas out

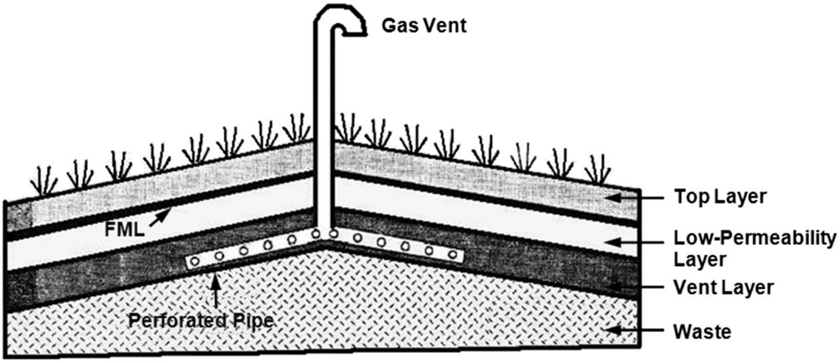


Figure 13. Passive Gas Control System (Venting to Atmosphere).[23]

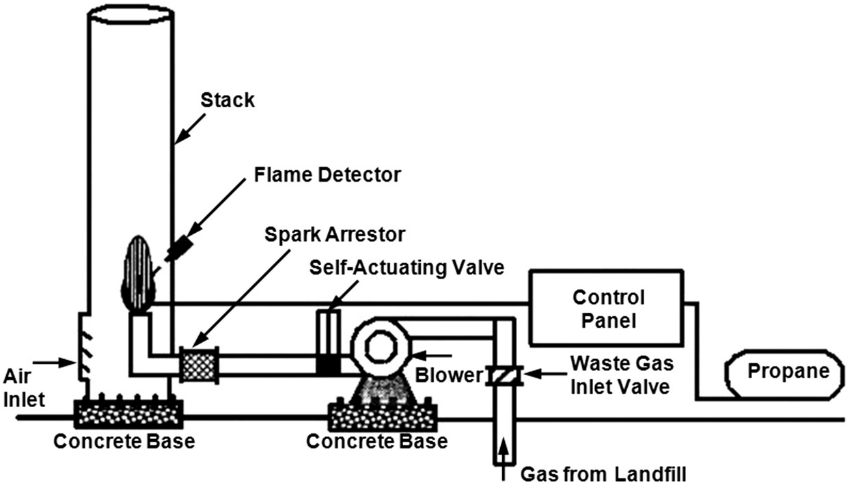


Figure 14. Schematic Diagram of a Ground-based Landfill Gas Flare.[23]

of the landfill. Negative pressure systems are more commonly used because they are more effective and offer more flexibility in controlling gas migration. The gas may be recovered for energy conversion, treated, or combusted in a flare system. Typical components of a flare system showed in Fig. 14.[23] Negative pressure systems may be used as either perimeter gas control systems or interior gas collection/recovery systems.

An active gas extraction well is depicted in Fig. 15[23] and Table 9[26] presents some guidelines for construction of vertical gas collection systems. Gas extraction wells may be installed within the landfill waste. A perimeter extraction trenches

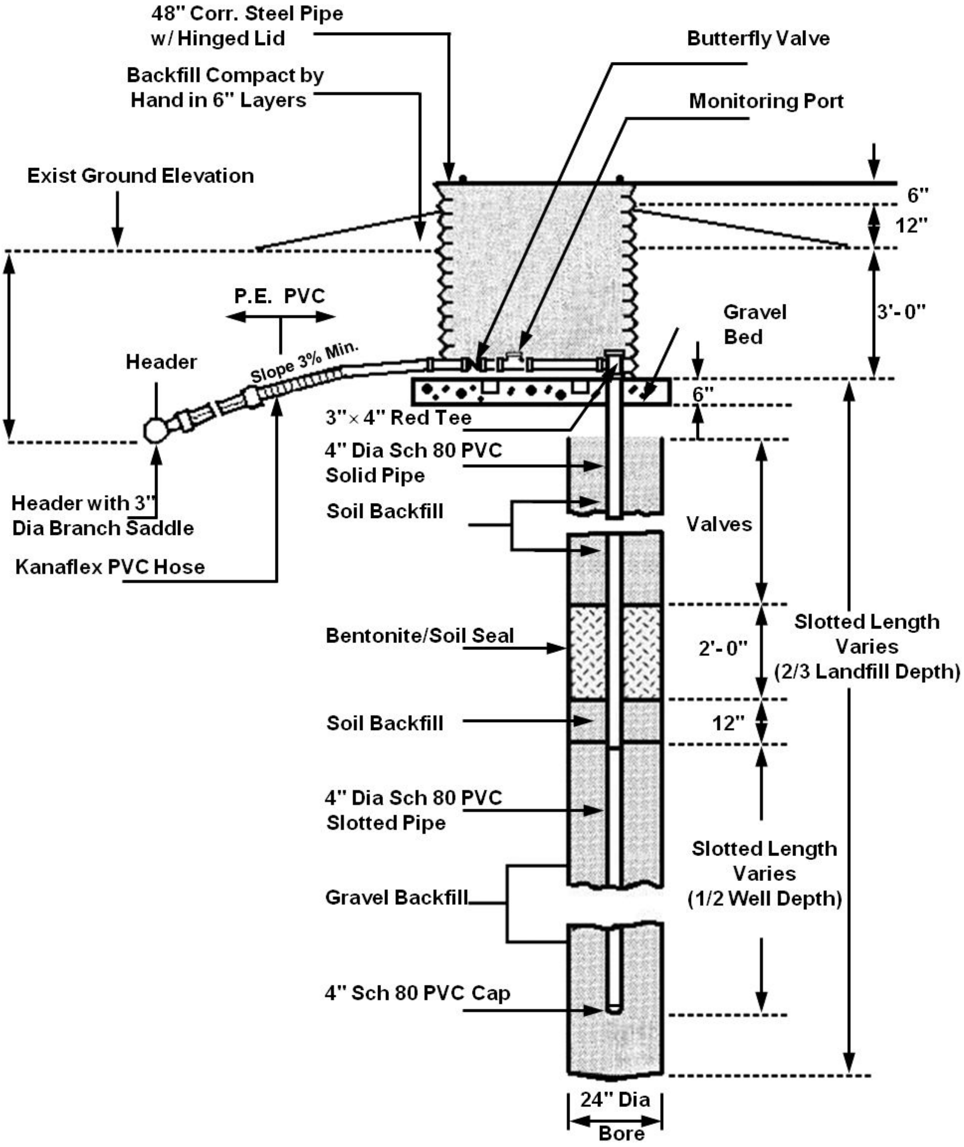


Figure 15. Schematic Illustration of a Gas Extraction Well.[23]

system is illustrated in Fig. 16.[23] One possible configuration of an interior gas collection/recovery system is illustrated in Fig. 17.[23]

The performance of active systems is not as sensitive to freezing or saturation of cover soils as that of passive systems. Although active gas systems are more effective in withdrawing gas from the landfill, capital, operation, and maintenance costs of such systems will be higher and these costs can be expected to continue throughout

Table 9. Guidelines for Construction of Vertical Gas Collection Systems.[26]

Parameter	Recommendation
Well depth	75% of depth or to water table, whichever come first
Perforations	Bottom 1/3 to 2/3
Casing	3–8 in. PVC or HDPE, telescoping well joint
Spacing (on center)	Interior collection system 200–500 ft, perimeter collection system 100–250 ft
Well density	One well per $\frac{1}{2}$ to 2 acres
Minimum gas collection piping slope	3%
Well bore diameter	12–36 in. is standard (24, 30, and 36 inches most common)

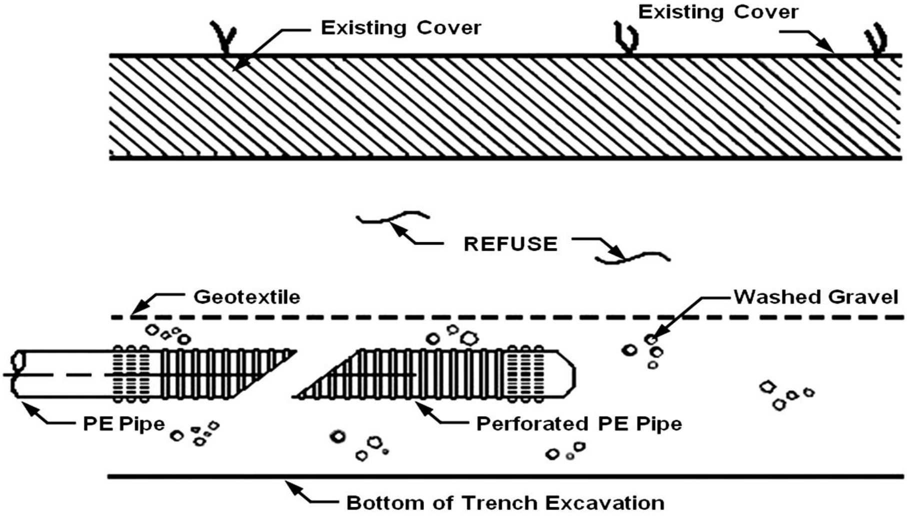


Figure 16. Schematic Diagram of a Perimeter Excavation Trench System.[23]

the post-closure period. At some future time, owners and operators may wish to convert active gas controls into passive systems when gas production diminishes. The conversion option and its environmental effect (i.e., gas release causing odors and health and safety concerns) should be addressed in the original design.

Suitability of the systems is based on the design and age of the landfill unit and on the soil, hydrogeologic, and hydraulic conditions of the facility and surrounding environment. Because of these variables, both systems have had varying degrees of success. Passive systems may be used in conjunction with active systems. An example of this may be the use of a low-permeability passive system for the closed portion of a landfill unit (for remedial purposes) and the installation of an active system in the active portion of the landfill unit (for future use).

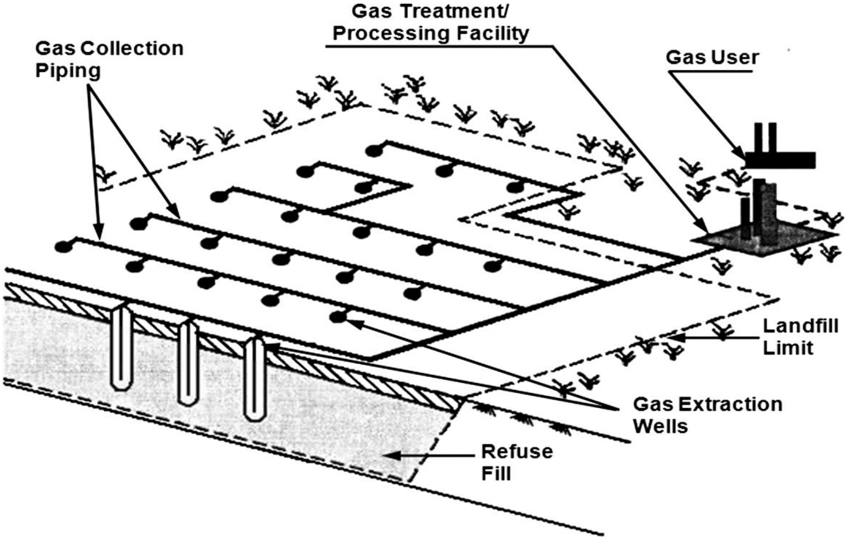


Figure 17. Schematic Illustration of An Interior Gas Collection/Recovery System.[23]

Selection of construction materials for either type of gas control system should consider the elevated temperature conditions within a landfill unit as compared to the ambient air or soil conditions in which gas control system components are constructed. Because ambient conditions are typically cooler, water-containing corrosive and possibly toxic waste constituents may be expected to condense. This condensate should be considered in selecting construction materials. Provisions for managing this condensate should be incorporated to prevent accumulation and possible failure of the collection system. The condensate can be returned to the landfill unit if the landfill is designed with a composite liner and LCS. The owners/ operators of certain landfill facilitated to install gas collection and control systems to reduce the emissions of nonmethane organic compounds (NMOCs). For new MSWLF units, as well as for those units that have a design capacity greater than 111,000 tons, a gas collection and control system must be installed if emissions evaluations indicate that the NMOC emissions rate is 150 MG/year (167 tons per year) or greater. Allowable control systems include open and enclosed flares, and on-site or off-site facilities that process the gas for subsequent sale or use.

3.6. *Air Monitoring*

Air pollution control requirements are essential prerequisite for construction of MSWLF units. The owner or operator of an MSWLF unit should consult the concern stakeholders/agencies those are responsible for air pollution control criteria. As per regulation of US EPA in United State, State Implementation Plan (SIP) of the Clean Air Act is the concern regulation for clean air criteria. The SIP may include variances, permits, or exemptions for burning agricultural wastes, silvicultural wastes, land-clearing debris, diseased trees, or debris from emergency clean-up operations. Routine burning of wastes is banned in all cases, and the SIP may limit burning of waste such as agricultural wastes to certain hours of the day; days of the year; designated burn areas; specific types of incinerators; atmospheric conditions; and distance from working face, public thoroughfares, buildings, and residences. Requirements under the SIP also may include notifying applicable state or local agencies whose permits may:

- restrict times when limited burning of waste may occur;
- specify periods when sufficient fire protection is deemed to be available; or
- limit burning to certain areas.

Open burning is defined as the combustion of solid waste:

- without control of combustion air to maintain adequate temperature for efficient combustion;
- without containment of the combustion reaction in an enclosed device to provide sufficient residence time and mixing for complete combustion; and
- without the control of the emission of the combustion products.

Trench or pit burners and air curtain destructors are considered open burning units, because the particulate emissions are similar to particulate emissions from open burning, and these devices do not control the emission of combustion products.

3.7. *Facility Access*

Owners and operators are required to control public access to prevent illegal dumping, public exposures to hazards at MSWLF units, and unauthorized vehicular traffic. Frequently, unauthorized persons are unfamiliar with the hazards associated with landfill facilities and consequences of uncontrolled access may include injury and even death. Potential hazards are related to inability of equipment operators to see unauthorized individuals during operation of equipment and haul vehicles; direct exposure to waste (i.e. sharp objects and pathogens); inadvertent or deliberate fires; and earth-moving activities. Acceptable measures used to limit access of unauthorized persons to the disposal facility include gates and fences, trees, hedges, berms, ditches, and embankments. Chain link, barbed wire added to chain link, and open farm-type fencing are examples of fencing that may be used. Access to facilities should be controlled through gates that can be locked when the site is unsupervised. Gates may be the only additional measure needed at remote facilities.

3.8. *Run-on/Run-off Control Systems*

Existing MSWLF units, lateral expansions, and new MSWLF unit owner or operator is required to prevent run-on onto the active portion of the landfill units and to collect and control run-off from the active portion for a 24-h operation, 25-year storm. Management of run-off must comply with the point and nonpoint source discharge requirements of the Clean Water Act (CWA) regulations in the United States.

If stormwater enters the landfill unit and contacts waste (including water within daily cover), the stormwater becomes leachate and must be managed as leachate. The purpose of a run-on control system is to collect and redirect surface waters to minimize the amount of surface water entering the landfill unit. Run-on control can be accomplished by constructing berms and swales above the filling area that will collect and redirect the water to stormwater control structures. Run-off control systems are designed to collect and control this run-off from the active portion of the landfill, including run-off from areas that have received daily cover, which may have contacted waste materials. Run-off control can be accomplished through stormwater conveyance structures that divert this run-off/leachate to the leachate storage device.

After a landfill unit has been closed with a final cover, stormwater run-off from this unit can be managed as stormwater and not as leachate. Therefore, waters running off the final cover system of closed areas may not require treatment and generally can be combined with run-on waters. For landfills with steep side slopes, a bench system may provide the best solution for run-off control. A bench creates a break in the slope where the velocity of the stormwater run-off is expected to become erosive. The bench converts sheet flow run-off into channel flow. Benches typically are spaced 30–50 ft apart up the slope. An alternative to benches is a system of down-chutes whereby stormwater is collected off the top of the landfill and conveyed down the slope through a pipe or channel. Caution should be taken not to construct downchutes with heavy material because of possible subsidence. Corrugated metal pipes or plastic-lined channels are examples of lightweight materials that can be used for downchute construction. Run-on and run-off control systems must be designed based on a 24-h operation, 25-year storm. Alternatively, local meteorological data can be analyzed to estimate the criterion storm. To estimate run-on, the local watershed should be identified and evaluated to document the basis for run-on design flows. The soil conservation service (SCS) method and/or the rational method are generally adequate for estimating storm flows for designing run-on and/or run-off control systems. The SCS method estimates run-off volume from accumulated rainfall and then applies the run-off volume to a simplified triangular unit hydrograph for peak discharge estimation and total run-off hydrograph. The rational method approximates the majority of surface water discharge supplied by the watershed upstream from the facility. The rational method generally is used for areas of less than 200 acres.

In the USA, run-on and run-off managed in accordance with the requirements of the CWA including, but not limited to, the National Pollutant Discharge Elimination System (NPDES). Run-on/run-off control structures, both temporary and permanent, may be incorporated into the system design. Other structures (not mentioned above) most frequently used for run-on/run-off control are waterways, seepage ditches, seepage basins, and sedimentation basins.

3.9. *Surface Water Requirements*

MSWLF units shall not cause a discharge of pollutants into waters including wetlands, which violates any requirements of the CWA including but not limited to the NPDES requirements in the USA. An MSWLF unit(s) that has a point source [*i.e. any discernible, confined, and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, landfill LCS, vessel or other floating craft from which pollutants are or may be discharged. This term does not include return flows from irrigated agriculture or agricultural storm water run-off*][27,28] discharge must have an NPDES permit. Point source discharges for landfills include, but are not limited to:

(1) the release of leachate from a leachate collection or on-site treatment system into the waters of the United States;
(2) disposal of solid waste into waters of the United States; or
(3) release of surface water (stormwater) run-off that is directed by a run-off control system into the waters.

Leachate that is piped or trucked off-site to a treatment facility is not regarded as a point source discharge. Owners/operators also should be aware that there are regulations promulgated pursuant to the CWA regarding stormwater discharges from landfill facilities. These regulations require stormwater discharge permit applications to be submitted by certain landfills, i.e. both uncontrolled and controlled sanitary landfills. "Uncontrolled sanitary landfills" are defined as landfills or open dumps that do not meet the requirements for run-on or run-off controls that are found in the MSWLF criteria. "Controlled sanitary landfills" are those that do meet the run-on and run-off requirements. Uncontrolled sanitary landfills owned or operated by municipalities of less than 100,000 (population) must submit for permit application for their stormwater discharge or obtain coverage under a general permit. For controlled sanitary landfills owned or operated by a municipality with a population less than 100,000, there is no requirement to submit a stormwater discharge permit application. However, a permit is required for municipalities with a population less than 250,000 that own or operate a municipal landfill.

3.10. *Liquids Restrictions*

The owner or operator is prohibited from placing bulk or noncontainerized liquid waste or containerized liquid waste into the MSWLF unit. Liquids from households are exempted whereas tank trucks of wastes are not. The restriction of bulk or containerized liquids is intended to control a source of liquids that may become a source of leachate. Liquid waste refers to any waste material that is determined to contain free liquids as defined by method of paint filter liquids test. The paint filter test is performed by placing a 100-ml sample of waste in a conical, 400-μm paint filter. The waste is considered a liquid waste if any liquid from the waste passes through the filter within 5-min. The apparatus used for performing the paint filter test is illustrated in Fig. 18.[23]

If the waste is considered a liquid waste, absorbent materials may be added to render a "solid" material (i.e., waste/absorbent mixture that no longer fails the paint filter liquids test). One common waste stream that may contain a significant quantity of liquid is sludge. Sludge is a mixture of water and solids that has been

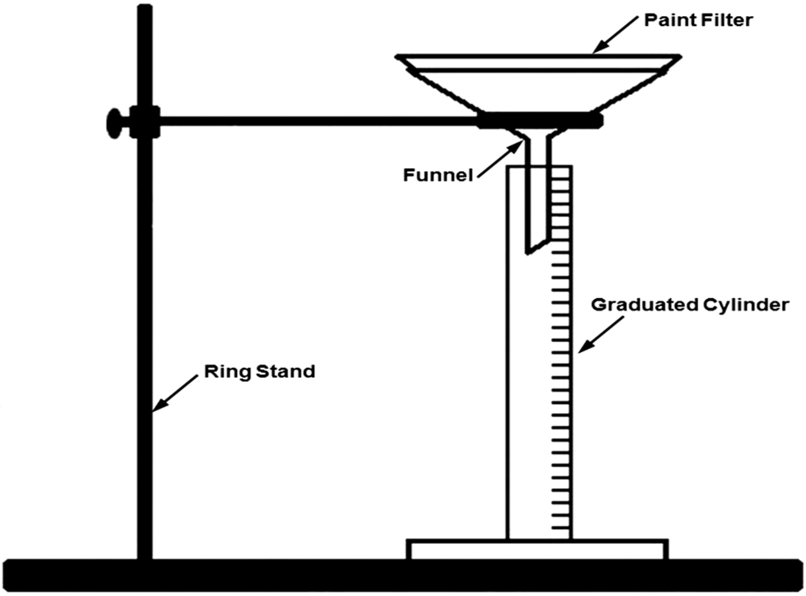


Figure 18. Schematic Arrangement of Paint Filter Test Apparatus.[23]

concentrated from, and produced during, water and wastewater treatments. Sludges may be produced because of providing municipal services (i.e., potable water supply, sewage treatment, storm drain maintenance) or commercial or industrial operations. Sewage sludge is a mixture of organic and inorganic solids and waters, removed from wastewater containing domestic sewage. Sludge disposal is acceptable provided the sludge passes the paint filter test.

Owners and operators of MSWLF units may return leachate and gas condensate generated from a gas recovery process to the MSWLF, provided the MSWLF unit has been designed and constructed with a composite liner and LCS. With some conditions, leachate and landfill gas condensate may allow recirculation in MSWLF units with alternative designs. Recirculating leachate or landfill gas concentrate may require demonstrating that the added volume of liquid will not increase the depth of leachate on the liner to more than 30 cm. Returning gas condensate to the landfill unit may represent a reasonable long-term solution for relatively small volumes of condensate. Gas condensate recirculation can be accomplished by pumping the condensate through pump stations at the gas recovery system and into dedicated drain fields (buried pipe) atop the landfill, or into other discharge points (i.e., wells). Because gas condensate may be odorous, spray systems for recirculation are not used unless combined with leachate recirculation systems.[23,28] Leachate recirculation to an MSWLF unit has been used as a measure for managing leachate or as a means of controlling and managing liquid and solid waste decompositions. Leachate recirculation can be accomplished in the same manner as recirculation of landfill gas condensate. Because of the larger volume, however, discharge points may not be as effective as drain-fields. In some cases, discharge points have been a source of odor. In addition, a discharge point may not allow for dissipation of the leachate.[28]

3.11. *Recordkeeping Requirements*

The owner or operator of an MSWLF unit must record and retain near the facility in an operating record or in an alternative location. The owner or operator is required to maintain records of demonstrations, inspections, monitoring results, design documents, plans, operational procedures, notices, cost estimates, and financial assurance documentation. The recordkeeping requirements are intended to be self-implementing of the MSWLF as per the regulations of the authorities. Recordkeeping at the landfill facility should include the following:

(a) Location restriction demonstrations are required for any location restrictions. The location restrictions apply to airports, floodplains, wetlands, fault areas, seismic impact zones, and unstable areas.

(b) Inspection records, training procedures, and notification procedures include: date and time wastes were received during the inspection; names of the transporter and the driver; source of the wastes; vehicle identification numbers; and all observations made by the inspector. Training records should include procedures used to train personnel on hazardous waste and on PCB waste recognition.

(c) Gas monitoring results and any remediation plans: if gas levels exceed 25% of the LEL for methane in any facility structures or exceed the LEL for methane at the facility boundary, the owner or operator must place in the operating record, within seven days, the methane gas levels detected, and a description of the steps taken to protect human health. Within 60 days of detection, the owner or operator must place a copy of the remediation plan used for gas releases in the operating record.

(d) MSWLF unit design documentation for placement of leachate or gas condensate in an MSWLF unit: if leachate and/or gas condensate are recirculated into the MSWLF unit, documentation of a composite liner and an LCS capable of maintaining a maximum of 30 cm of leachate head in the MSWLF unit must be placed in the operating record.

(e) Demonstration, certification, monitoring, testing, or analytical finding required by the groundwater criteria: documents to be placed in the operating record include: (i) documentation of design, installation, development, and decommission of any monitoring wells, piezometers, and other measurement, sampling, and analytical devices; (ii) certification by a qualified groundwater scientist of the number, spacing, and depths of the monitoring systems; (iii) documentation of sampling and analysis programs and statistical procedures; (iv) notice of finding a statistically significant increase over background for one or more of the constituents at any monitoring well at the waste management unit boundary or the relevant point of compliance; (v) certification by experts that an error in sampling, analysis, statistical evaluation, or natural variation in groundwater caused an increase (false positive) the level of the contamination; (vi) a notice identifying any constituents that have been detected in groundwater and their concentrations; (vii) a notice identifying the constituents that have exceeded the GWPS; (viii) a certification by experts that a source other than the MSWLF unit caused the contamination or an error in sampling, analysis, statistical evaluation, or natural groundwater variation caused a statistically significant increase (false positive); (ix) the remedies selected to remediate groundwater contamination; and (x) certification of remediation completion.

(f) Closure and post-closure plans and any monitoring, testing, or analytical data associated with these plans: the landfill facility owner or operator is required to place a copy of the closure plan, post-closure plan, and a notice of intent to close the facility in the operating record. Monitoring, testing, or analytical data associated with closure and post-closure information generated from groundwater and landfill gas monitoring must be placed in the operating record. A copy of the notation on the deed to the MSWLF facility property, as required following closure, along with certification and verification that closure and post-closure activities have been completed in accordance with their respective plans, also must be placed in the operating record.

(g) Estimates and financial assurance documentation required. The following documents must be placed in the operating record: (i) an estimate of the cost of hiring a third party to close the largest area of all MSWLF units that will require final cover; (ii) justification for the reduction of the closure cost estimate and the amount of financial assurance (if appropriate); (iii) a cost estimate of hiring a third party to conduct post-closure care; (iv) the justification for the reduction of the post-closure cost estimate and financial assurance (if appropriate); (v) an estimate and financial assurance for the cost of a third party to conduct corrective action, if necessary; and (vi) a copy of the financial assurance mechanisms.

4. Design Criteria

4.1. *General Overview*

New MSWLF facilities and lateral expansions of existing units must comply with either a design standard or a performance standard for landfill design. The design standard requires a composite liner composed of two feet of soil with a hydraulic conductivity of no more than 1×10^{-7} cm/s, overlain by an FML and an LCS. A performance-based design must demonstrate the capability of maintaining contaminant concentrations below maximum contaminant levels at the unit's relevant point of compliance. The performance standard has been established to allow design innovation and consideration of site-specific conditions; however, different location may have adopted alternative design standards according to their local conditions. Owners/operators are advised to work closely with permitting agencies to determine the applicable design standard.

The design considerations discussed in this chapter are intended to identify the key design features and system components for the composite liner and LCS standards, and for the performance standard. The technical considerations included (1) design concepts, (2) design calculations, (3) physical properties, and (4) construction methods for the following considerations:

Designs based on the performance standard included:

- Leachate characterization and leakage assessment;
- Leachate migration and migration models; and
- Relevant point of compliance assessment.

Composite liners and LCSs included:

- Soil liner component (soil properties laboratory testing, design, construction, and quality assurance (QA)/quality control (QC) testing);
- FMLs (properties, design installation, and QA/QC testing);
- LCSs (strength and compatibility, grading and drainage, clogging potential, and filtration);
- Leachate removal systems (pumps, sumps, and standpipes); and
- Inspections (field observations and field and laboratory testing).

The following sub-sections address the minimum regulatory requirements that should be considered during the design, construction, and operation of MSWLF units to ensure that they perform in a manner protective of human health and the environment. Additional features or procedures may be used to demonstrate conformance with the regulations or to control leachate release and subsequent effects. For example, during construction of a new MSWLF unit, or a lateral expansion of an existing MSWLF unit, QC and QA procedures and documentation may be used to ensure that material properties and construction methods meet the design specifications that are intended to achieve the expected level of performance.[29]

4.2. *Design Based on the Performance Standard*

There are any design not conforming to the uniform design standard of a composite liner system and an LCS for MSWLFs, therefore; the owner or operator of the proposed MSWLF unit must demonstrate to the concern authorities that the design will not allow the maximum concentration limit of contaminant chemicals in groundwater at the relevant point of compliance. The demonstration should consider an assessment of leachate quality and quantity, leachate leakage to the subsurface, and subsurface transport to the relevant point of compliance. These factors are governed by site hydrogeology, waste characteristics, and climatic conditions. Table 10[29] presents the maximum contamination limit (MCL) of chemicals in the groundwater at relevant point of compliance of an MSWLF.

The nature of the demonstration is essentially an assessment of the potential for leachate production and leakage from the landfill to groundwater, and the anticipated fate and transport of constituents listed in Table 10 to the proposed relevant point of compliance at the facility. Inherent in this approach is the need to evaluate whether contaminants in groundwater at the relevant point of compliance will exceed the concentration values listed in Table 10. If so, then the owner or operator needs to obtain sufficient site-specific data to adequately characterize the existing groundwater quality and the existing groundwater flow regime (e.g., flow

Table 10. MCL of Chemicals in the Groundwater at Relevant Point of Compliance of an MSWLF.[29]

Chemical Name	MCL (mg/L)	Chemical Name	MCL (mg/L)
Arsenic	0.05	Lead	0.05
Barium	1.0	Mercury	0.002
Benzene	0.005	Mythoxychlor	0.1
Cadmium	0.01	Nitrate	10.0
Carbon tetrachloride	0.005	Selenium	0.01
Chromium (hexavalent)	0.05	Silver	0.05
2,4-Dichlorophenoxy acetic acid	0.1	Toxaphene	0.005
1,4-Dichlorobenzene	0.075	1,1,1-Trichloroethane	0.2
1,2-Dichloroethane	0.005	Trichloroethylene	0.005
1,1-Dichloroethylene	0.007	2,4,5-Trichlorophenoxy acetic acid	0.01
Endrin	0.0002	Vinyl chloride	0.002
Fluoride	4.0		
Lindane	0.004		

direction, horizontal and vertical gradients, hydraulic conductivity, stratigraphy, and aquifer thickness).

An assessment should be made of the effect MSWLF facility construction will have on site hydrogeology. The assessment should focus on the reduced infiltration over the landfill area and altered surface water run-off patterns. Reduction of groundwater recharge and changes in surface water patterns resulting from landfill construction may affect groundwater gradients in some cases and may result in changes in lateral flow directions. One example of a hypothetical performance-based demonstration follows. It is possible that an MSWLF unit located in an arid climatic zone would not produce leachate from sources of water (i.e. precipitation) other than that existing within the waste at the time of disposal. In such an environment, an owner or operator may demonstrate that significant quantities of leachate would not be produced. The demonstration should be supported by evaluating historic precipitation and evaporation data and the likelihood that the unit could be flooded because of heavy rains, surface run-off, or high water tables. It may be possible, through operational controls, to avoid exposing waste to precipitation or infiltration of water through overlying materials. If significant leachate production would not be expected, the regulatory authority, when reviewing the demonstration, should consider the hydrogeologic characteristics of the facility and the surrounding area, in addition to the expected volume of leachate and climatic factors. Assuming leachate is produced, the demonstration should evaluate whether or not constituents listed in Table 10 can be expected to be present at concentrations greater than the MCLs. If such a demonstration is possible, it must address the hydrogeologic characteristics of the facility and the surrounding land. The types of wastewater generated in individual landfill in the United States are presented in Table 11.[28]

Table 11. Wastewater Flows Generated by Individual Landfills in the USA.[28]

Type of Wastewater Generated	Number of Landfills	Minimum Average Flow (gal/day)	Maximum Average Flow (gal/day)	Industry Median (gal/day)
Floor washing	70	10	5,450	743
Landfill leachate	1,989	1	533,000	5,620
Contaminated groundwater	163	6	987,000	12,800
Stormwater run-off	1,135	10	2,067,000	26,800
Landfill gas condensate	158	3	11,700	343
Recovering pumping wells	50	0.3	80,200	136
Truck/equipment washwater	416	5	15,000	118
Drained free liquids	33	1	82,000	253
Other	2	0	0	0
Total	4,016			

4.2.1. *Leachate Characterization and Leakage Assessment*

4.2.1.1. Leachate characterization

Leachate characterization should include an assessment and demonstration of the quantity and composition of leachate anticipated at the proposed facility. Estimates of volumetric production rates of leachate are important in evaluating the fate and transport of the constituents listed in Table 10. Leachate production rates depend on rainfall, run-on, run-off, evapotranspiration, water table elevation relative to the bottom of the landfill unit, in-place moisture content of waste, and the prevention of liquid disposal at the site. Run-on, run-off, and water table factors can be managed traditionally through design and operational controls. The MSWLF criteria prohibit bulk or containerized liquid disposal. Incident precipitation and evapotranspiration can be evaluated using models (i.e., HELP — hydraulic evaluation of landfill performance) or other methods of estimating site-specific leachate production (i.e., local historical meteorological data). If leachate composition data that are representative of the proposed facility are not available, then leachate data with a similar expected composition should be presented. Landfill leachate composition is influenced by:

- The annual infiltration of precipitation and rate of leaching;
- The type and relative amounts of materials in the waste stream; and
- The age and the biological maturity of the landfill unit, which may affect the types of organic and inorganic acids generated, oxidation/reduction potential (Eh), and pH conditions.

An existing landfill unit in the same region, with similar waste stream characteristics, may provide information that will allow the owner or operator to anticipate leachate

Table 12. Leachate Characteristics at MSWLF with Varying Age in the USA.[28]

	MSWLF Age Group (Year in Which Landfill Facility Began Accepting Waste), Parameters Presented as Median Concentration		
Pollutant Parameter (unit)	Pre — 1980	1980 — 1990	1990 — Present (2000)
Ammonia (mg/L)	140	95	48
BOD (mg/L)	210	125	344
COD (mg/L)	596	930	3,038
TOC (mg/L)	445	377	150
TSS (mg/L)	202	49	100
Alpha terpineol (μg/L)	746	123	—
Benzoic acid (μg/L)	75	9,308	—
P-Cresol (μg/L)	25	117	—
Phenol (μg/L)	17	242	820
Chromium (μg/L)	27	31	10
Zinc (μg/L)	145	93	139

composition of the proposed landfill unit. Table 12[28] presents leachate characteristics at MSWLF with varying age in the USA.

A wide range of leachate concentrations are reported in the literature with higher concentrations of specific constituents typically reported for the initial leachate from laboratory or field experimental test fills or test cells. These “batch” one-day landfill tests do not account for the long-term climatic and meteorological influences on a full-scale landfill operation. Such high initial concentrations are not typical of full-scale operations (which are subject to the dilution effects of incidental rainfall on unused portions of the unit).

4.2.1.2. Assessment of leakage through liners

An assessment of leakage (the volumetric release of leachate from the proposed performance-based design) should be based on analytical approaches supported by empirical data from other existing operational facilities of similar design, particularly those that have leak detection monitoring systems. In lieu of the existence or availability of such information, conservative analytical assumptions may be used to estimate anticipated leakage rates.

The transport of fluids and waste constituents through geomembranes differs in principle from transport through soil liner materials. The dominant mode of leachate transport through liner components is flow through holes and penetrations of the geomembrane, and *Darcian flow* through soil components. Transport through geomembranes where tears, punctures, imperfections, or seam failures are not involved is dominated by molecular diffusion. Diffusion occurs in response to a concentration gradient and is governed by *Fick's first law*. Diffusion rates through geomembranes are very low in comparison to hydraulic flow rates in soil liners, including compacted clays. For synthetic liners, the most significant factor influencing liner performance is penetration of the liner, including imperfect seams or pinholes caused by construction defects in the geomembrane.

A relatively new product now being used in liner systems is the geosynthetic clay liner (GCL). GCLs consist of a layer of pure bentonite clay backed by one or two geotextiles. GCLs exhibit properties of both soil liners and geomembranes and have successfully substituted for the soil component in composite liner designs. GCLs are believed to transport fluids primarily through diffusion according to their low hydraulic conductivities (i.e. 1×10^{-9} cm/s reported by manufacturers). Applications for GCLs are discussed further in the sections that follow. Table 13 presents the flow through different liners.[30,31]

Several researchers have studied the flow of fluids through imperfections in single geomembrane and composite liner systems. Leakage assessments also may be conducted with the use of the HELP model; however, the current version of the model is under revision and will include an updated method to assess leakage that is based on recent research and data.

4.2.2. *Leachate Migration and Migration Models*

4.2.2.1. Leachate migration in the subsurface

Leachate that escapes from a landfill unit may migrate through the unsaturated zone and eventually reach the uppermost aquifer. In some instances, however, the water table may be located above the base of the landfill unit, so that only saturated flow and transport from the landfill unit need to be considered. Once leachate reaches the water table, contaminants may be transported through the saturated zone to a point of discharge (i.e. a pumping well, a stream, a lake, etc.). The migration of leachate in the subsurface depends on factors such as the volume of the liquid component of the waste, the chemical and physical properties of the leachate constituents, the loading

Table 13. Flow Through Different Liners.[30,31]

Liner Quality and Type	Holes Per Acre	Rate of Flow (gal/ac/day)	Rate of Flow (L/ha/day)
Good FML	1 @ $0.1\,\mathrm{cm}^2$	330	3,100
Excellent clay	1×10^{-8} cm/s[a]	12	110
Poor composite	30 @ $0.1\,\mathrm{cm}^2$	19	180
Poor composite	1 @ $1\,\mathrm{cm}^2$	0.8	7
Excellent FML	none	0.01	0.1

[a] Hydraulic conductivity.

rate, climate, and the chemical and physical properties of the subsurface (saturated and unsaturated zones). A number of physical, chemical, and biological processes also may influence migration. Complex interactions between these processes may result in specific contaminants being transported through the subsurface at different rates. Certain processes result in the attenuation and degradation of contaminants. The degree of attenuation is dependent on the time that the contaminant is in contact with the subsurface material, the physical and chemical characteristics of the subsurface material, the distance that the contaminant has traveled, and the volume and characteristics of the leachate. Some of the key processes affecting leachate migration are discussed briefly in the following.

4.2.2.1.1. *Physical processes controlling contaminant transport in the subsurface*

Physical processes that control the transport of contaminants in the subsurface include advection, mixing and dilution because of dispersion and diffusion, mechanical filtration, physical sorption, multi-phase fluid flow, and fracture flow. These processes, in turn, are affected by hydrogeologic characteristics, such as hydraulic conductivity and porosity, and by chemical processes.

- *Advection* — It is the process by which solute contaminants are transported by the overall motion of flowing groundwater. A nonreactive solute will be transported at the same rate and in the same direction as groundwater flow. Advective transport is chiefly a function of the subsurface hydraulic conductivity distribution, porosity, and hydraulic gradients.
- *Hydrodynamic dispersion* — It is a nonsteady and irreversible mixing process by which a contaminant plume spreads as it is transported through the subsurface. Dispersion results from the effects of two components operating at the microscopic level: mechanical dispersion and molecular diffusion. Mechanical dispersion results from variations in pore velocities within the soil or aquifer and may be more significant than molecular diffusion in environments where the flow rates are moderate to high. Molecular diffusion occurs because of contaminant concentration gradients; chemicals move from high concentrations to low concentrations. At very slow groundwater velocities, as occur in clays and silts, diffusion can be an important transport mechanism.
- *Mechanical filtration* — It removes from groundwater contaminants that are larger than the pore spaces of the soil. Thus, the effects of mechanical filtration increase with decreasing pore size within a medium. Filtration occurs over a wide range of particle sizes. The retention of larger particles may effectively reduce the permeability of the soil or aquifer.

- *Physical sorption* — It is a function of *van der Waals forces*, and the hydrodynamic and electrokinetic properties of soil particles. Sorption is the process by which contaminants are removed from solution in groundwater and adhere or cling to a solid surface. The distribution of a contaminant between the solution and the solid phase is called partitioning.
- *Multiphase fluid flow* — It occurs because many solvents and oils are highly insoluble in water and may migrate in the subsurface as a separate liquid phase. If the viscosity and density of a fluid differ from that of water, the fluid may flow at a different rate and direction than the groundwater. If the fluid is denser than water, it may reach the bottom of the aquifer (top of an aquitard) and may alter its flow direction to conform to the shape and slope of the aquitard surface.
- *Hydraulic conductivity* — It is a measure of the ability of geologic media to transmit fluids. It is a function of the size and arrangement of water-transmitting openings (pores and fractures) in the media and of the characteristics of the fluids (density, viscosity, etc.). Spatial variations in hydraulic conductivity are referred to as heterogeneities. A variation in hydraulic conductivity with the direction of measurement is referred to as anisotropy.
- *Variable hydraulic conductivity of the geologic formation* — It may cause groundwater flow velocities to vary spatially. Variations in the rate of advection may result in nonuniform plume spreading. The changes in aquifer properties that lead to this variability in hydraulic conductivity may be 3D. If the geologic medium is relatively homogeneous, it may be appropriate, in some instances, to assume that the aquifer properties also are homogeneous.
- *Secondary porosity in rock* — It may be caused by the dissolution of rock or by regional fracturing; in soils, secondary porosity may be caused by desiccation cracks or fissures. Fractures or macropores respond quickly to rainfall events and other fluid inputs and can transmit water rapidly along unexpected pathways. Secondary porosity can result in localized high concentrations of contaminants at significant distances from the facility. The relative importance of secondary porosity to hydraulic conductivity of the subsurface depends on the ratio of fracture hydraulic conductivity to inter-granular hydraulic conductivity. For scenarios in which fracture flow is dominant, the relationships used to describe porous flow (*Darcy's Law*) do not apply.

4.2.2.1.2. *Chemical processes controlling contaminant transport in the subsurface*

Chemical processes that are important in controlling subsurface transport include precipitation/dissolution, chemical sorption, redox reactions, hydrolysis, ion exchange, and complexation. In general, these processes, except for hydrolysis, are reversible. The reversible processes tend to retard transport, but do not permanently remove a contaminant from the system. Sorption and precipitation are generally the dominant mechanisms retarding contaminant transport in the saturated zone.

- *Precipitation/dissolution* — This reaction can control contaminant concentration levels. The solubility of a solid controls the equilibrium state of a chemical. When the soluble concentration of a contaminant in leachate is higher than that of the equilibrium state, precipitation occurs. When the soluble concentration is lower than the equilibrium value, the contaminant exists in solution. The precipitation of a dissolved substance may be initiated by changes in pressure, temperature, pH, concentration, or redox potential. Precipitations of contaminants in the pore space of an aquifer can decrease aquifer porosity. Precipitation and dissolution reactions are especially important processes for trace metal migration in soils.
- *Chemical adsorption/desorption* — It is the most common mechanism affecting contaminant migration in soils. Solutes become attached to the solid phase by means of adsorption. Like precipitation/dissolution, adsorption/desorption is a reversible process. However, adsorption/desorption generally occurs at a relatively rapid rate compared to precipitation reactions. The dominant mechanism of organic sorption is the hydrophobic attraction between a chemical and natural organic matters that exist in some aquifers. The organic carbon content of the porous medium and the solubility of the contaminant are important factors for this type of sorption. There is a direct relationship between the quantity of a substance sorbed on a particle surface and the quantity of the substance suspended in solution. Predictions about the sorption of contaminants often make use of sorption isotherms, which relate the amount of contaminant in solution to the amount adsorbed to the solids. For organic contaminants, these isotherms are usually assumed to be linear and the reaction is assumed to be instantaneous and reversible. The linear equilibrium approach to sorption may not be adequate for all situations.
- *Oxidation and reduction* (redox) — These reactions involve the transfer of electrons and occur when the redox potential in leachate is different from that of the soil or aquifer environment. Redox reactions are important processes for inorganic compounds and metallic elements. Together with pH, redox reactions affect the solubility, complexing capacity, and sorptive behavior of constituents, and thus control the presence and mobility of many substances in water. Microorganisms are responsible for a large proportion of redox reactions that occur in groundwater. The redox state of an aquifer and the identity and quantity of redox-active reactants are difficult to determine.
- *Hydrolysis* — It is the chemical breakdown of carbon bonds in organic substances by water and its ionic species H^+ and OH^-. Hydrolysis is dependent on pH and

activity of electrons (Eh) and is most significant at high temperatures, low pH, and low redox potential. For many biodegradable contaminants, hydrolysis is slow compared to biodegradation.

- *Ion exchange* — It originates primarily from exchange sites on layered silicate clays and organic matter that have a permanent negative charge. Cation exchange balances negative charges in order to maintain neutrality. The capacity of soils to exchange cations is called the cation exchange capacity (CEC). CEC is affected by the type and quantity of clay mineral present, the amount of organic matter present, and the pH of the soil. Major cations (i.e. Ca, Mg, K, Na, etc.) in leachate usually dominate the CEC sites, resulting in little attenuation in soils of trace metals in the leachate. A smaller ion exchange effect for anions is associated with hydrous oxides. Soils typically have more negatively charged clay particles than positively charged hydrous oxides. Therefore, the transport of cations is attenuated more than the transport of anions.
- *Complexation* — It involves reactions of metal ions with inorganic anions or organic ligands. The metal and the ligand bind together to form a new soluble species called a complex. Complexation can either increase the concentration of a constituent in solution by forming soluble complex ions or decrease the concentration by forming a soluble ion complex with a solid. It is often difficult to distinguish among sorption, solid–liquid complexation, and ion exchange. Therefore, these processes are usually grouped together as one mechanism.

4.2.2.1.3. *Biological processes controlling contaminant transport in the subsurface*

Biodegradation of contaminants may result from the enzyme-catalyzed transformation of organic compounds by microbes. Contaminants can be degraded to harmless byproducts or to more mobile and/or toxic products through one or more of several biological processes. Biodegradation of a compound depends on environmental factors such as redox potential, dissolved oxygen concentration, pH, temperature, presence of other compounds and nutrients, salinity, depth below land surface, competition among different types of organisms, and concentrations of compounds and organisms. The transformations that occur in a subsurface system are difficult to predict because of the complexity of the chemical and biological reactions that may occur. Quantitative predictions of the fate of biologically reactive substances are subject to a high degree of uncertainty, in part, because little information is available on biodegradation rates in soil systems or groundwater. First-order decay constants are often used instead.

The operation of Subtitle D facilities as per US EPA can introduce bacteria and viruses into the subsurface. The fate and transport of bacteria and viruses in the

Table 14. Types of Wastewater Treatment Employed by the Landfills Industry in the USA.[28]

	Number of Landfills in the USA		
Type of Treatment	Direct Discharge	Indirect Discharge	Zero Discharge
No treatment	81	691	468
Biological treatment	119	37	19
Chemical precipitation	63	45	8
Chemical precipitation and biological treatment	32	10	0
Filtration and biological treatment	45	4	5
Equalization and biological treatment	65	28	7
Equalization, biological treatment, and filtration	37	4	5
Equalization, chemical precipitation, and biological treatment	26	8	0
Equalization, chemical precipitation, biological treatment, and filtration	26	2	0

subsurface is an important consideration in the evaluation of the effects of MSWLF units on human health and the environment. A large number of biological, chemical, and physical processes are known to influence viral and bacterial survivals and transport in the subsurface. Unfortunately, the knowledge of the processes and the available data are insufficient to develop models that can simulate a wide variety of site-specific conditions. Table 14 shows different types of wastewater treatment employed by the landfills industry in the USA.[28]

4.2.2.2. Leachate migration models

After reviewing the hydrogeologic characteristics of the site, the nature of liner leakage, and the leachate characteristics, it may be appropriate to use a mathematical model to simulate the expected fate and transport of the constituents listed in Table 10 to the relevant point of compliance. Solute transport and groundwater modeling efforts should be conducted by an expert groundwater hydrologist. It is necessary to consider several factors when selecting and applying a model to a site.

4.2.2.2.1. *Overview of the modeling process*

A number of factors can influence leachate migration from MSWLF units. These include, but are not limited to, climatic effects, the hydrogeologic setting, and the nature of the disposed waste. Each facility is different and no one generic model will be appropriate in all situations. To develop a model for a site, the modeling needs and the objectives of the study should be determined first. Next, it will be necessary to collect data to characterize the hydrological, geological, chemical, and biological conditions of the system. These data are used to assist in the development of a conceptual model of the system, including spatial and temporal characteristics and boundary conditions. The conceptual model and data are then used to select a mathematical model that accurately represents the conceptual model. The model selected should have been tested and evaluated by qualified investigators, should adequately simulate the significant processes present in the actual system, and should be consistent with the complexity of the study area, amount of available data, and objectives of the study.

First, an evaluation of the need for modeling should be made (Fig. 19).[29,32] When selecting a model to evaluate the potential for soil and groundwater contamination, three basic determinations must be made (Fig. 20).[29,32] This decision

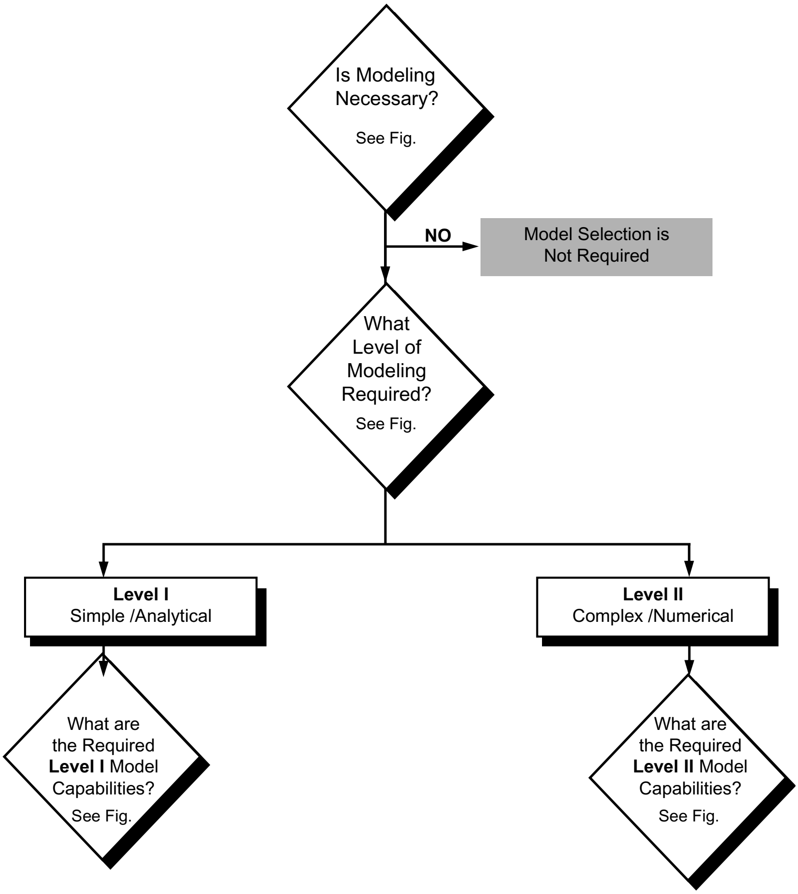


Figure 19. Three Basic Decisions in Model Selection.[29,32]

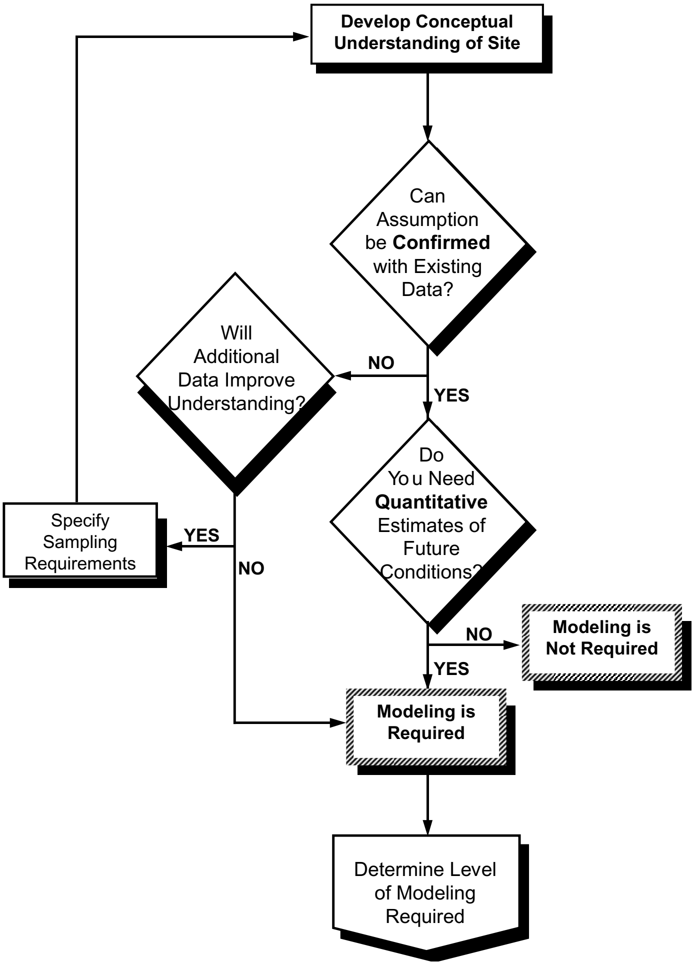


Figure 20. Flowchart for Determining the Modeling Requirement.[29,32]

should be made at the beginning of the study, since modeling may require a substantial amount of resources and effort. Next, the level of model complexity required for a specific study should be determined (Fig. 21).[29,32] Models classify as Level I (simple/analytical) and Level II (complex/numerical) models. A flowchart for determining the level of model complexity required is shown in Fig. 21. Finally, the model capabilities necessary to represent a particular system should be considered (Fig. 22).[29,32]

Several models may be equally suitable for a particular study. In some cases, it may be necessary to link or couple two or more computer models to accurately represent the processes at the site. In the section that follows, specific issues that should

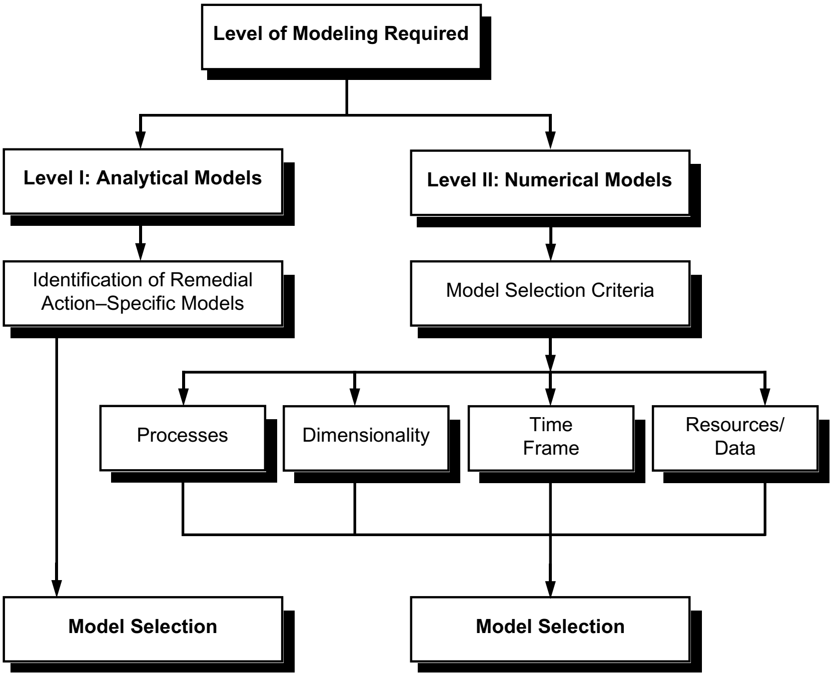


Figure 21. Flowchart to Determine the Level of Modeling Required for Soil and Groundwater Systems.[29,32]

be considered when developing a scenario and selecting a model are described. Models are a simplified representation of the real system, and as such, cannot fully reproduce or predict all site characteristics. Errors are introduced as a result of:

(1) simplifying assumptions;
(2) a lack of data;
(3) uncertainty in existing data;
(4) a poor understanding of the processes influencing the fate and transport of contaminants; and
(5) limitations of the model itself.

Therefore, model results should be interpreted as estimates of groundwater flow and contaminant transport. However, models could be used for comparing various scenarios, since all scenarios would be subject to the same limitations and simplifications. The quality of model results can depend to a large extent on the experience and judgment of the modeler and on the quality of the data used to develop model input. The process of applying the model may highlight data deficiencies that may require additional data collection. The model results should be calibrated to obtain

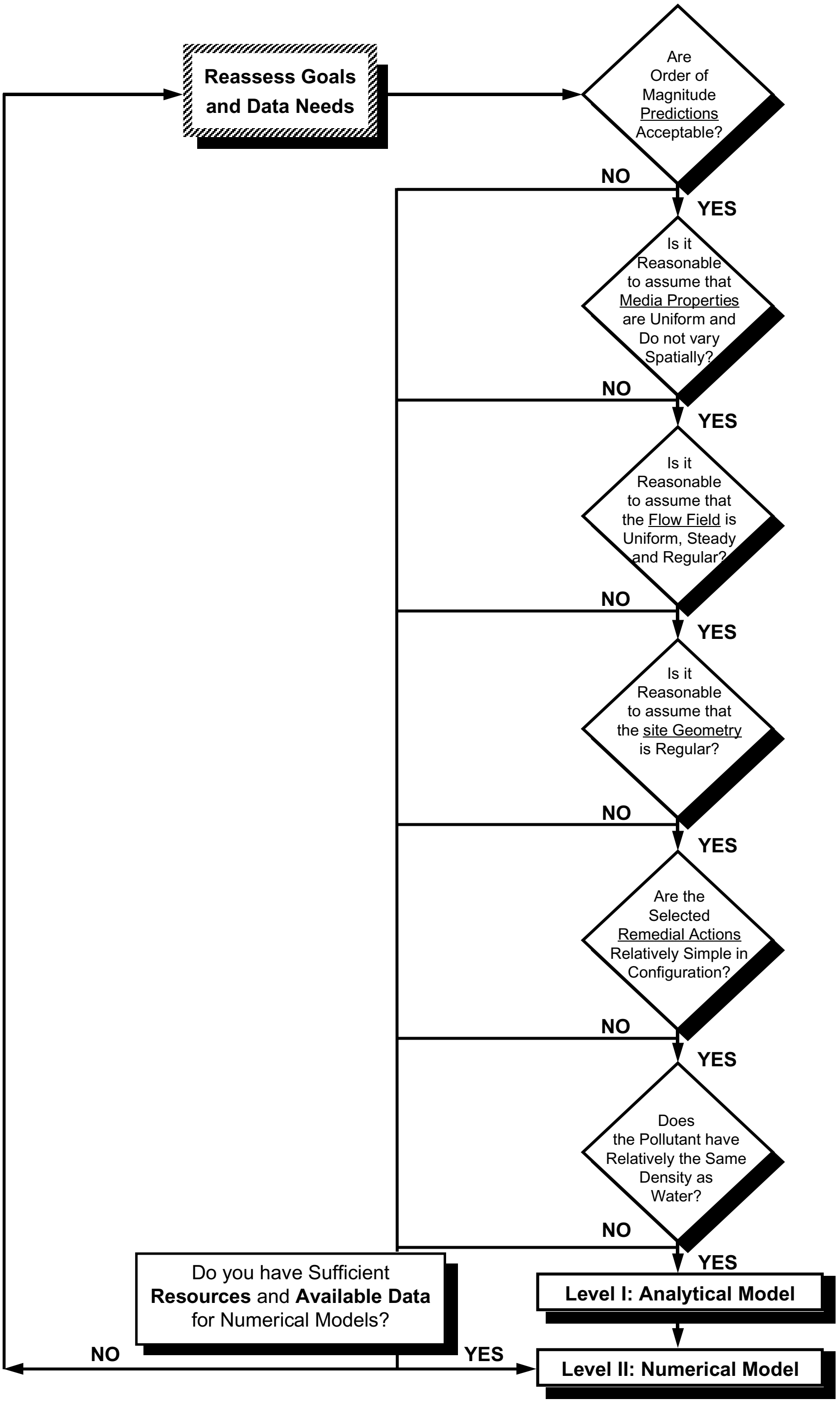


Figure 22. Flowchart for Required Model Capabilities for Soil and Groundwater Systems.[29,32]

the best fit to the observed data. The accuracy of the results obtained from modeling efforts should then be validated. Model validation, which is the comparison of model results with experimental data or environmental data, is a critical aspect of model application and is particularly important for site-specific evaluations.

Several recent reports present detailed discussions of the issues associated with model selection, application, and validation. In the United States, EPA's Exposure Assessment Group (US EPA–EAG) has developed suggested recommendations and guidance on model validation. A recent report by the National Research Council (NRC) discusses the issues related to model application and validation, and provides recommendations for the proper use of groundwater models.

4.2.2.3. Model selection considerations

To develop a true "worst-case" model, information on the probable heterogeneity and anisotropy of the site should be groundwater flow and solute transport models range from simple and analytical calculations to sophisticated computer programs that use numerical solutions to solve mathematical equations describing flow and transport. A sophisticated model may not yield an exact estimate of water quality at the relevant point of compliance for a given set of site conditions, but it may allow an estimate of the effects of complex physical and chemical processes. Depending on the complexity of site conditions and the appropriateness of the simplifying assumptions, a fairly sophisticated numerical model may provide useful estimates of water quality at the relevant point of compliance. The following considerations should be addressed when selecting a model.

4.2.2.3.1. *Analytical versus numerical models*

Mathematical models use either analytical, semi-analytical, or numerical solutions for groundwater flow and transport equations. Each technique has advantages and disadvantages. Analytical solutions are computationally more efficient than numerical simulations and are more conducive to uncertainty analysis (i.e., Monte Carlo techniques). Typically, input data for analytical models are simple and do not require detailed familiarity with the computer model or extensive modeling experience. Analytical solutions are typically used when data necessary for characterization of the site are sparse and simplifying assumptions are appropriate.[33] The limited data available in most field situations may not justify the use of a detailed numerical model; in some cases, results from simple analytical models may be appropriate. Analytical models require simplifying assumptions about the system. Therefore, complex interactions involving several fate and transport processes cannot be addressed in detail. Analytical models generally require a limited number of parameters that are often assumed to be constant in space and time.

Semi-analytical models approximate complex analytical solutions using numerical techniques. Semi-analytical methods allow for more complex site conditions than those that can be simulated with a purely analytical solution. Semi-analytical solution methods can consider multiple sources or recharging and discharging wells. However, they still require simplifying assumptions about the dimensionality and homogeneity of the system. Numerical models are able to evaluate more complex site conditions than either analytical or semi-analytical models. Numerical models provide the user with a large amount of flexibility; irregular boundaries and spatial and temporal variations in the system can be considered. Numerical models require significantly more data than analytical models and they are typically more computationally intensive. Use of a numerical model requires an experienced modeler and it can involve a larger amount of computer time than simulations using an analytical or semi-analytical method.

To select an appropriate model, the complexity of the site hydrology and the availability of data should be considered. If data are insufficient, a highly sophisticated and complex model should not be used. In some situations, it is beneficial to use an analytical or semi-analytical model as a “screening-level” model to define the range of possible values and to use a numerical model when there are sufficient data. A highly complex hydrogeologic system cannot be accurately represented with a simple analytical model. Heterogeneous or anisotropic aquifer properties, multiple aquifers, and complicated boundary conditions can be simulated using numerical models. In addition, sophisticated numerical models are available that can simulate processes such as fracture flow. Because each site is unique, the modeler should determine which conditions and processes are important at a specific site, and then select a suitable model.

4.2.2.3.2. *Spatial characteristics of the system*

Although actual landfill units and hydrogeologic systems are 3D, it is often desirable to reduce the number of dimensions simulated in a mathematical model to one or two. 2D and 3D models are generally more complex and computationally expensive than 1D models and, therefore, require more data. In some instances, a 1D model may adequately represent the system; the available data may not warrant the use of a multi-dimensional model. However, modeling a truly 3D system using a 2D model may produce results without adequate spatial detail. The choice of the number of dimensions in the model should be made for a specific site, based on the complexity of the site and the availability of data.

4.2.2.3.3. *Steady-state versus transient models*

Models can simulate either steady-state or transient flow conditions. It may be appropriate to assume that some groundwater flow systems have reached approximate "steady-state" conditions, which implies that the system has reached equilibrium and no significant changes are occurring over time. The assumption of steady-state conditions generally simplifies the mathematical equations used to describe flow processes and reduces the amount of input data required.

However, assuming steady-state conditions in a system that exhibits transient behavior may produce inaccurate results. For example, climatic variables, such as precipitation, vary over time and may have strong seasonal components. In such settings, the assumption of constant recharge of the groundwater system would be incorrect. Steady-state models also may not be appropriate for evaluating the transport of chemicals that sorb or transform significantly. The choice of simulating steady-state or transient conditions should be based on the degree of temporal variability in the system.

4.2.2.3.4. *Boundary and initial conditions*

The solution of differential equations describing flow and transport processes requires that initial and boundary conditions be specified. The initial conditions describe the conditions present in the system at the beginning of the simulation. In many groundwater flow and transport models, these conditions are related to the initial hydraulic conditions in the aquifer and the initial concentration of contaminants. Boundary conditions define the conditions present on the borders of the system, which may be steady-state or temporally variable. The initial and boundary conditions chosen to represent a site can significantly affect the results of the simulation. One of the most significant boundary conditions in solute transport models is the introduction of a contaminant to the system. A source of groundwater contamination should be described in terms of its spatial, chemical, and physical characteristics, and its temporal behavior. Spatially, a source may be classified as a point source, line source, a distributed source of limited areal or 3D extent, or as a nonpoint source of unlimited. Typically, temporal descriptions of the source term boundary conditions for models with analytical solutions are constant, constant pulse, and/or exponential decay. Numerical models typically handle a much wider range of source boundary conditions, allowing for a wide range of contaminant loading scenarios.

4.2.2.3.5. *Homogeneous versus heterogeneous aquifer/soil properties*

The extent of the spatial variability of the properties of each aquifer will significantly affect the selection of a mathematical model. Many models assume uniform aquifer properties, which simplifies the governing equations and improves computational efficiency. For example, a constant value of hydraulic conductivity may be assumed at every point in the aquifer. However, this assumption may ignore the heterogeneity in the hydrogeologic system. When site-specific data are limited, it is common to assume homogeneous and isotropic aquifer properties and to develop a "reasonable worst-case" scenario for contaminant migration in the subsurface. However, the assumption of homogeneous and isotropic aquifers often will not provide a "worst-case" scenario. For example, a continuous zone of higher hydraulic conductivity in the direction of groundwater flow can result in much higher rates of contaminant movement than would be predicted in a completely homogeneous aquifer. The number of aquifers in the hydrogeologic system also will affect the selection of a mathematical model. Some systems include only a single unconfined or confined aquifer, which is hydraulically isolated from the surrounding layers. Some mathematical models, and in particular those with analytical solutions, can simulate only single layers. In other cases, the upper aquifer may be hydraulically connected to underlying aquifers. The MSWLF Criteria specify that MCLs not be exceeded at the relevant point of compliance within the uppermost aquifer. The uppermost aquifer includes not only the aquifer that is nearest the ground surface, but also all lower aquifers that are hydraulically connected to the uppermost aquifer within the vicinity of the facility.

4.2.2.3.6. *Availability of data*

Although computer models can be used to make predictions about leachate generation and migration, these predictions are highly dependent on the quantity and quality of the available data. One of the most common limitations to modeling is insufficient data. Uncertainty in model predictions results from the inability to characterize a site in terms of the boundary conditions or the key parameters describing the significant flow and transport processes. The application of a mathematical model to a site typically requires a large amount of data. Inexperienced modelers may attempt to apply a model with insufficient data and, as a result, produce model results that are inconclusive.[34]

To obtain accurate model results, it is essential to use data that are appropriate for the particular site being modeled. Models that include generic parameters, based on average values for similar sites, can be used to provide initial guidance and general information about the behavior of a system, but it is inappropriate to apply generic parameters to a specific hydrogeologic system. Mercer *et al.*[35] studied an excellent summary of the data required to model saturated and unsaturated flow, surface water flow, and solute transport. This report provides definitions and possible ranges of values for source terms, dependent variables, boundary conditions, and initial conditions.

4.3. *Composite Liners and LCS*

New MSWLF units and expansions of existing MSWLF units in states without approved programs must be constructed with a composite liner and an LCS that is designed to maintain a depth of leachate less than 30 cm (12 in.) above the liner. A composite liner consists of an FML installed on top of, and in direct and uniform contact with, two feet of compacted soil. The FML must be at least 30-mil thick unless the FML is made of high-density polyethylene (HDPE), which must be 60-mil thick. The compacted soil liner must be at least two-feet thick and must have a hydraulic conductivity of no more than 1×10^{-7} cm/sec. The FML component must be installed in direct and uniform contact with the compacted soil component. This sub-section provides information on the components of composite liner systems including soils, geomembranes, and LCSs.

4.3.1. *Standard Composite Liner Systems*

The composite liner system is an effective hydraulic barrier because it combines the complementary properties of two different materials into one system:

- Compacted soil with a low-hydraulic conductivity ($\geq 1 \times 10^{-7}$ cm/sec) and
- An FML (FMLs are also referred to as geomembranes).

Geomembranes may contain defects including tears, improperly bonded seams, and pinholes. In the absence of an underlying low-permeability soil liner, flow through a defect in a geomembrane is essentially unrestrained. The presence of a low-permeability soil liner beneath a defect in the geomembrane reduces leakage by limiting the flow rate through the defect.

Flow through the soil component of the liner is controlled by the size of the defect in the geomembrane, the available air space between the two liners into which leachate can flow, the hydraulic conductivity of the soil component, and the hydraulic head. Fluid flow through soil liners is calculated by *Darcy's Law*, where discharge (Q) is proportional to the head loss through the soil (dh/dl) for a given cross-sectional flow area (A) and hydraulic conductivity (K) where:

$$Q = KA(dh/dl) \tag{1}$$

Leakage through a geomembrane without defects is controlled by Fick's first law, which describes the process of liquid diffusion through the membrane liner. The diffusion process is similar to flow governed by Darcy's law for soil liners except that diffusion is driven by concentration gradients and not by hydraulic head. Although diffusion rates in geomembranes are several orders of magnitude lower than comparable hydraulic flow rates in low-permeability soil liners, construction of a completely impermeable geomembrane is difficult. The factor that most strongly influences geomembrane performance is the presence of imperfections such as improperly bonded seams, punctures, and pinholes. A detailed discussion of leakage through geomembranes and composite liners can be found in the documents of US EPA.[30,31] A geomembrane installed with excellent control over defects may yield the equivalent of a one-centimeter-diameter hole per acre of liner installed (Table 13). If the geomembrane were to be placed over sand, this size imperfection under one foot of constant hydraulic head could be expected to account for as much as 3,300 gal/acre/day (31,000 liters/hectare/day) of leakage. Based upon measurements of actual leakage through liners that have been built under rigorous control at facilities have estimated an actual leakage rate, under one foot of constant head, of 200 liters/hectare/day or about 21 gallons/acre/day for landfill units.

The uniformity of the contact between the geomembrane and the soil liner is extremely important in controlling the effective flow area of leachate through the soil liner. Porous material, such as drainage sand, filter fabric, or other geofabric, should not be placed between the geomembrane and the low permeability soil liner. Porous materials will create a layer of higher hydraulic conductivity, which will increase the amount of leakage below an imperfection in the geomembrane. Construction practices during the installation of the soil and the geomembrane affect the uniformity of the geomembrane/soil interface, and strongly influence the performance of the composite liner system.

4.3.1.1. Soil liner

The following subsections discuss soil liner construction practices including thickness requirements, lift placement, bonding of lifts, test methods, prerequisite soil properties, QC, and QA activities.

4.3.1.1.1. *Thickness*

Two feet of soil is generally considered the minimum thickness needed to obtain adequate compaction to meet the hydraulic conductivity requirement. This thickness is considered necessary to minimize the number of cracks or imperfections through the entire liner thickness that could allow leachate migration. Both lateral and vertical imperfections may exist in a compacted soil. The two-foot minimum thickness is believed to be sufficient to inhibit hydraulic short-circuiting of the entire layer.

4.3.1.1.2. *Lift thickness*

Soil liners should be constructed in a series of compacted lifts. Determination of appropriate lift thickness is dependent on the soil characteristics, compaction equipment, firmness of the foundation materials, and the anticipated compactive effort needed to achieve the required soil hydraulic conductivity. Soil liner lifts should be thin enough to allow adequate compactive effort to reach the lower portions of the lift. Thinner lifts also provide greater assurance that sufficient compaction can be achieved to provide good and homogeneous bonding between subsequent lifts. Adequate compaction of lift thickness between five and ten inches is possible if appropriate equipment is used. Nine-inch loose lift thicknesses that will yield a six-inch soil layer also have been recommended prior to compaction. Soil liners usually are designed to be of uniform thickness with smooth slopes over the entire facility. Thicker areas may be considered wherever recessed areas for leachate collection pipes or collection sumps are located. Extra thickness and compactive efforts near edges of the side slopes may enhance bonding between the side slopes and the bottom liner. In smaller facilities, a soil liner may be designed for installation over the entire area, but in larger or multi-cell facilities, liners may be designed in segments. If this is the case, the design should address how the old and new liner segments will be bonded together.

4.3.1.1.3. *Bonding between lifts*

It is not possible to construct soil liners without some microscopic and/or macroscopic zones of higher and lower hydraulic conductivity. Within individual lifts, these preferential pathways for fluid migration are truncated by the bonded zone between the lifts. If good bonding between the lifts is not achieved during construction, the vertical pathways may become connected by horizontal pathways at the lift interface, thereby diminishing the performance of the hydraulic barrier. Two methods may be used to ensure proper bonding between lifts. Kneading or blending a thinner, new lift with the previously compacted lift may be achieved by using a footed roller with long feet that can fully penetrate a loose lift of soil. If the protruding rods or feet of a sheep's-foot roller are sufficient in length to penetrate the top lift and knead the previous lift, good bonding may be achieved. Another method includes scarifying (i.e. roughening), and possibly wetting, the top inch or so of the last lift placed with a disc harrow or other similar equipment before placing the next lift.

4.3.1.1.4. *Placement of soil liners on slopes*

The method used to place the soil liner on side slopes depends on the angle and length of the slope. Gradual inclines from the toe of the slope enable continuous placement of the lifts up the slopes and provide better continuity between the bottom and sidewalls of the soil liner. When steep slopes are encountered, however, lifts may need to be placed and compacted horizontally due to the difficulties of operating heavy compaction equipment on steeper slopes. When sidewalls are compacted horizontally, it is important to tie in the edges with the bottom of the soil liner to reduce the probability of seepage planes. A significant amount of additional soil liner material will be required to construct the horizontal lifts since the width of the lifts has to be wide enough to accommodate the compaction equipment. After the soil liner is constructed on the side slopes using this method, it can be trimmed back to the required thickness. The trimmed surface of the soil liner should be sealed by a smooth-drum roller. The trimmed excess materials can be reused provided that they meet the specified moisture-density requirements.

4.3.1.1.5. *Hydraulic conductivity*

Achieving the hydraulic conductivity standard depends on the degree of compaction, compaction method, type of clay, soil moisture content, and density of the soil during liner construction. Hydraulic conductivity is the key design parameter when evaluating the acceptability of the constructed soil liner. The hydraulic conductivity of a soil depends, in part, on the viscosity and density of the fluid flowing through it. While water and leachate can cause different test results, water is an acceptable fluid for testing the compacted soil liner and source materials. The effective porosity of the soil is a function of size, shape, and area of the conduits through which the liquid flows. The hydraulic conductivity of a partially saturated soil is less than the hydraulic conductivity of the same soil when saturated. Because invading water only flows through water-filled voids (and not air-filled voids), the dryness of a soil tends to lower permeability. Hydraulic conductivity testing should be conducted on samples that are fully saturated to attempt to measure the highest possible hydraulic conductivity. There were several methods to measure the hydraulic conductivity of soil samples [*i.e. US EPA method 9100, Test Methods for Evaluating Solid Waste (SW 846); U.S. Army Corps of Engineers Engineering Manual 1110-2-1906 (COE, 1970) and Measurement of Hydraulic Conductivity of Saturated Porous Materials Using a Flexible Wall Permeameter (ASTM D-5084)*]. To verify full saturation of the sample, the American Society for Testing and Materials (ASTM) method may be performed with back pressure saturation and electronic pore pressure measurement.

4.3.1.1.6. *Soil properties*

Soils typically possess a range of physical characteristics, including particle size, gradation, and plasticity, that affect their ability to achieve a hydraulic conductivity of $\geq 1 \times 10^{-7}$ cm/sec. Testing methods used to characterize proposed liner soils should include grain size distribution (ASTM D-422), *Atterberg limits* (ASTM D-4318), and compaction curves depicting moisture and density relationships using the standard or modified Proctor (ASTM D-698 or ASTM D-1557), whichever is appropriate for the compaction equipment used and the degree of firmness of the foundation materials.

Liner soils usually have at least 30% fines (fine silt- and clay-sized particles). Some soils with less than 30% fines may be worked to obtain hydraulic conductivities below 1×10^{-7} cm/sec, but the use of these soils requires greater control of construction practices and conditions. The soil plasticity index (PI), which is determined from the *Atterberg limits* (defined by the liquid limit minus the plastic limit), should generally be $>10\%$. However, soils with very high PI (greater than 30%) are cohesive and sticky and become difficult to work within the field. When high PI soils are too dry during placement, they tend to form hard clumps (clods) that are difficult to break down during compaction. Preferential flow paths may be created around the clods allowing leachate to migrate at a relatively high rate. Soil particles or rock fragments also can create preferential flow paths. For this reason, soil particles or rock fragments should be less than three inches in diameter so as not to affect the overall hydraulic performance of the soil liner. The maximum density of a soil will be achieved at the optimum water content, but this point generally does not correspond to the point at which minimum hydraulic conductivity is achieved. Wet soils, however, have low shear strength and high potential for desiccation cracking. Care should be taken not to compromise other engineering properties such as shear strengths of the soil liner by excessively wetting the soil liner. Depending on the specific soil characteristics, compaction equipment and compactive effort, the hydraulic conductivity criterion may be achieved at moisture values of 1–7% above the optimum moisture content.

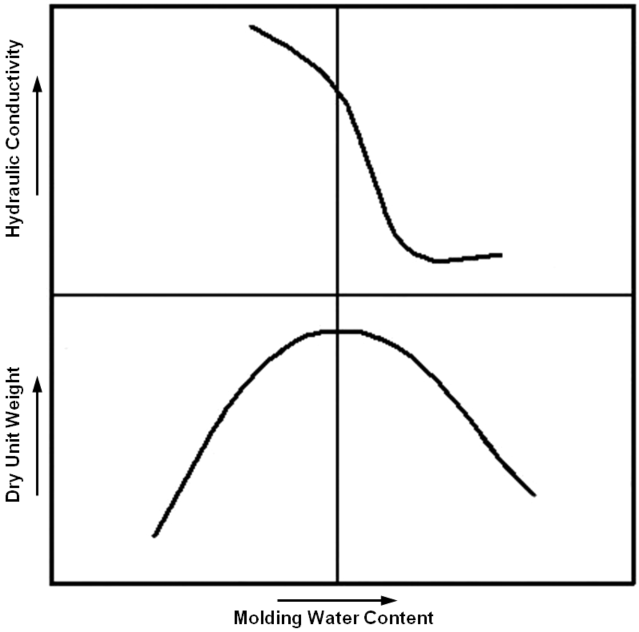
Although the soil may possess the required properties for successful liner construction, the soil liner may not meet the hydraulic conductivity criterion if the construction practices used to install the liner are not appropriate and carefully controlled. Construction QC and QA will be discussed in a later section.

4.3.1.1.7. *Amended soils*

If locally available soils do not possess properties to achieve the specified hydraulic conductivity, soil additives can be used. Soil additives, such as bentonite or other clay materials, can decrease the hydraulic conductivity of the native soil. Bentonite may be obtained in a dry and powdered form that is relatively easy to blend with on-site soils. Bentonite is a clay mineral (sodium-montmorillonite) that expands when it comes into contact with water (hydration), by absorbing the water within the mineral matrix. This property allows relatively small amounts of bentonite (5–10%) to be added to a noncohesive soil (sand) to make it more cohesive. Thorough mixing of additives to cohesive soils (clay) is difficult and may lead to inconsistent results with respect to complying with the hydraulic conductivity criterion. The most common additive used to amend soils is sodium bentonite. The disadvantage of using sodium bentonite includes its vulnerability to degradation as a result of contact with chemicals and waste leachates. Calcium bentonite, although more permeable than sodium bentonite, also is used as a soil amendment. Approximately twice as much calcium bentonite typically is needed to achieve a hydraulic conductivity comparable to that of sodium bentonite. Soil/bentonite mixtures generally require central plant mixing by means of a pugmill, cement mixer, or other mixing equipment where water can be added during the process. Water, bentonite content, and particle size distribution must be controlled during mixing and placement. Spreading of the soil/bentonite mixture may be accomplished in the same manner as the spreading of natural soil liners, by using scrapers, graders, bulldozers, or a continuous asphalt paving machine. Materials other than bentonite, including lime, cement, and other clay minerals such as atapulgite, may be used as soil additives.

4.3.1.1.8. *Testing*

Prior to construction of a soil liner, the relationship among water content, density, and hydraulic conductivity for a particular soil should be established in the laboratory. Figure 23[29] showed the influence of molding water content (i.e. moisture content of the soil at the time of compaction) on hydraulic conductivity of the soil. The lower half of the diagram is a compaction curve and shows the relationship between dry unit weight, or dry density of the soil, and water content of the soil. The optimum



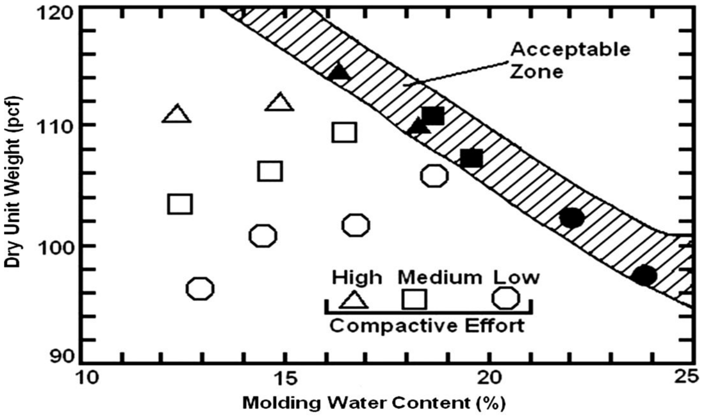
Note: The optimum moisture content occurs at the point at which maximum density is achieved. The lowest hydraulic conductivity generally occurs at water contents higher than optimum.

Figure 23. Schematic Illustration Represent the Hydraulic Conductivity and Dry Unit Weight as a Function of Molding Water Content.[29]
Note: The optimum moisture content occurs at the point at which maximum density is achieved. The lowest hydraulic conductivity generally occurs at water contents higher than optimum.

moisture content of the soil is related to a peak value of dry density known as maximum dry density. Maximum dry density is achieved at the optimum moisture content.

The lowest hydraulic conductivity of compacted clay soil is achieved when the soil is compacted at a moisture content that is slightly higher than the optimum moisture content, generally in the range of 1–7%. When compacting clay, water content and compactive effort are the two factors those should be controlled to meet the maximum hydraulic conductivity criterion. It is impractical to specify and construct a clay liner to a specific moisture content and a specific compaction (i.e. 5% wet of optimum and 95% modified Proctor density). Moisture content can be difficult to control in the field during construction; therefore, it may be more appropriate to specify a range of moisture contents and corresponding soil densities (percentage compaction) that are considered appropriate to achieve the required hydraulic conductivity. Figure 24[29] presented compaction data as a function of dry unit weight and molding water content for the construction of clay liners. The amount of soil testing required to determine these construction parameters is dependent on the degree of natural variability of the source material.

QA and QC of soil liner materials involve both laboratory and field testings. QC tests are performed to ascertain compaction requirements and the moisture content of material delivered to the site. Field tests for QA provide an opportunity to check representative areas of the liner for conformance to compaction specifications, including density and moisture content. QA laboratory testing is usually conducted



Note: Solid symbols represent specimens with a hydraulic conductivity $\leq 1 \times 10^{-7}$ cm/s and open symbols represent specimens with a hydraulic conductivity $> 1 \times 10^{-7}$ cm/s.

Figure 24. Schematic Represent of Compaction for Silty Clay Soil.[29]
Note: Solid symbols represent specimens with a hydraulic conductivity $\leq 1 \times 10^{-7}$ cm/s and open symbols represent specimens with a hydraulic conductivity $> 1 \times 10^{-7}$ cm/s.

on field samples for determination of hydraulic conductivity of the in-place liner. Laboratory testing allows full saturation of the soil samples and simulates the effects of large overburden stress on the soil, which cannot be done conveniently in the field. Differences between laboratory and field conditions (i.e. uniformity of material, control of water content, compactive effort, and compaction equipment) may make it unlikely that minimum hydraulic conductivity values measured in the laboratory on remolded, pre-construction borrow source samples are the same as the values achieved during actual liner construction. Laboratory testing on remolded soil specimens does not account for operational problems that may result in desiccation, cracking, poor bonding of lifts, and inconsistent degree of compaction on sidewalls.

In situ, or field, hydraulic conductivity testing operates on the assumption that by testing larger masses of soil in the field, one can obtain more realistic results. Four types of *in situ* hydraulic conductivity tests generally are used: (i) borehole tests, (ii) porous probes, (iii) infiltrometer tests, and (iv) under-drain tests. A borehole test is conducted by drilling a hole, then filling the hole with water, and measuring the rate at which water percolates into the borehole. In the borehole test, water also can percolate through the sidewalls of the borehole. As a result, the measured hydraulic conductivity is usually higher than that measured by other 1D field testing. The second type of test involves driving or pushing a porous probe into the soil and pouring water through the probe into the soil. With this method, however, the advantage of testing directly in the field is somewhat offset by the limitations of testing such a small volume of soil. A third method of testing involves a device called an infiltrometer. This device is embedded into the surface of the soil liner such that the rate of flow of a liquid into the liner can be measured. The two types of infiltrometers most widely used are open and sealed. Open rings are less desirable because, with a hydraulic conductivity of 10^{-7} cm/sec, it is difficult to detect a 0.002 inch per day drop in water level of the pond from evaporation and other losses. With sealed rings, very low rates of flow can be measured. However, single-ring infiltrometers allow lateral flow beneath the ring, which can complicate the interpretation of test results. Single rings are also susceptible to the effects of temperature variation; as the water temperature increases, the entire system expands. As it cools down, the system contracts. This situation could lead to erroneous measurements when the rate of flow is small. The sealed double-ring infiltrometer has proven to be the most successful method and is the one currently used. The outer ring forces infiltration from the inner ring to be more or less 1D. Covering the inner ring with water insulates it substantially from temperature variation. Underdrains, the fourth type of *in situ* test, are the most accurate *in situ* permeability testing device because they measure exactly what migrates from the bottom of the liner. However, underdrains are slow to generate data for low permeability liners, because of the length of time required to accumulate measurable flow. In addition, underdrains must be installed during construction, so fewer underdrains are used than other kinds of testing devices.

Other than the four types of field hydraulic conductivity tests, by ASTM: D 2937 [*"Standard Test Method for Density of Soil in Place by the Drive-Cylinder Method"*] may be used to obtain in-place hydraulic conductivity of the soil liner. This test method uses a surface soil sampler to drive a thin-walled cylinder (typically $3'' \times 3''$ inch^2) into a completed lift of the soil liner to obtain relatively undisturbed samples for laboratory density and hydraulic conductivity testing. This test can provide useful correlation to other field and QA testing results (i.e. *Atterberg limits*, gradation, in-place moisture, and density of the soil liner) to evaluate the in-place hydraulic conductivity of the soil liner.

Field hydraulic conductivity tests are not usually performed on the completed liner because the tests may take several weeks to complete (during which time the liner may be damaged by desiccation or freezing temperatures) and because large penetrations must be made into the liner. If field conductivity tests are performed, they are usually conducted on a test pad. The test pad should be constructed using the materials and methods to be used for the actual soil liner. The width of a test pad is usually the width of three to four construction vehicles and the length is one to two times the width. Thickness is usually 2–3 ft. Test pads can be used as a means for verifying that the proposed materials and construction procedures will meet performance objectives. If a test pad is constructed, if tests verify that performance objectives have been met, and if the actual soil liner is constructed to standards that equal or exceed those used in building the test pad (as verified through QA), then the actual soil liner should meet or exceed performance objectives.

4.3.1.1.9. *Soil liner construction*

Standard compaction procedures are usually employed when constructing soil liners. The following factors influence the degree and quality of compaction:

- lift thickness;
- full-scale or segmented lift placement;
- number of equipment passes;
- scarification between lifts;
- soil water content; and
- the type of equipment and compactive effort.

The method used to compact the soil liner is an important factor in achieving the required minimum hydraulic conductivity. Higher degrees of compactive effort increase soil density and lower the soil hydraulic conductivity for a given water content. The results of laboratory compaction tests do not necessarily correlate directly with the amount of compaction that can be achieved during construction. Heavy compaction equipment (>25,000 lbs or 11,300 kg) is typically used when building the soil liner to maximize compactive effort. The preferred field compaction equipment is a sheep's-foot roller with long feet that fully penetrates loose lifts of soil and provides higher compaction while kneading the clay particles together. The shape and depth of the feet are important; narrow and rod-like feet with a minimum length of about 7 in. provide the best results. A progressive change from the rod-like feet to a broader foot may be necessary in some soils after initial compaction, to allow the roller to walk out of the compacted soil. The sheep's-foot feet also aid in breaking up dry clods discussed earlier in subsection as soil properties. Mechanical road reclaimers, which are typically used to strip and repave asphalt, can be extremely effective in reducing soil clod size prior to compaction and in scarifying soil surfaces between lifts. Other equipment that has been used to compact soil includes discs and rototillers. To achieve adequate compaction, the lift thickness (usually 5–9 in.) may be decreased or the number of passes over the lift may be increased. Generally, compaction equipment should pass over the soil liner 5–20 times to attain the compaction needed to comply with the minimum hydraulic conductivity criterion. Efforts made to reduce clod size during excavation and placement of the soil for the liner should improve the chances for achieving low-hydraulic conductivity in several ways. Keeping clods in the soil liner material small will facilitate more uniform water content. Macropores between clod remnants can result in unacceptably high-field-hydraulic conductivity. Opinions differ on acceptable clod sizes in the uncompacted soil. Some suggest a maximum of 1–3 in. in diameter, or no larger than one-half the lift thickness. The main objective is to remold all clods in the compaction process to keep hydraulic conductivity values consistent throughout the soil liner.

4.3.1.2. Geomembranes and or synthetic liner

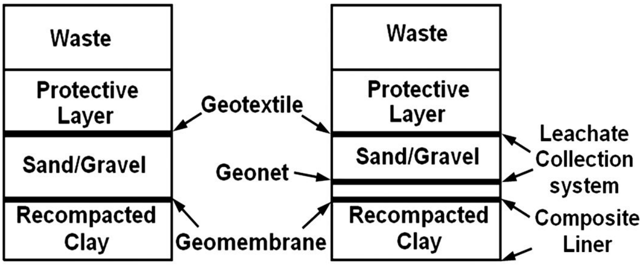
Geomembranes are called as FML. Geomembranes are relatively thin sheets of flexible thermoplastic or thermoset polymeric materials that are manufactured and prefabricated at a factory and transported to the site. Because of their inherent impermeability, the use of geomembranes in landfill unit construction has increased. Some other types of synthetic liners are as follows:

Geotextiles — In landfill liners, geotextiles are used to prevent the movement of small soil and refuse particles into the leachate collection layers and to protect geomembranes from punctures. These materials allow the movement of water but trap particles to reduce clogging in the LCS.

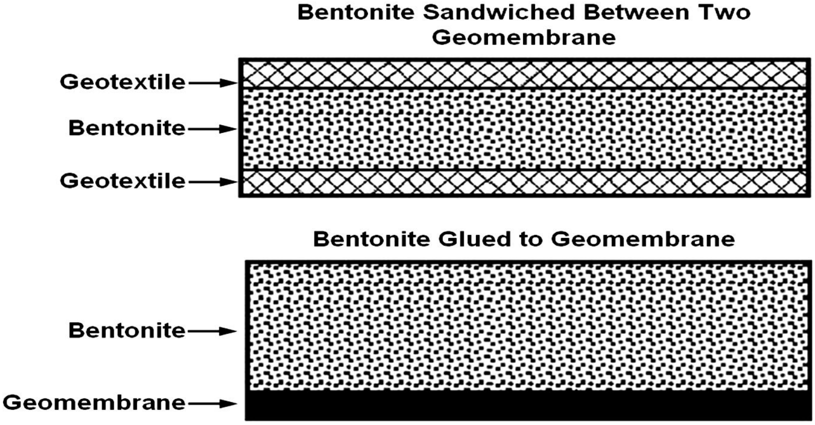
Geosynthetic Clay Liner — GCLs are becoming more common in landfill liner designs. These liners consist of a thin clay layer (4–6 mm) between two layers of a geotextile. These liners can be installed more quickly than traditional compacted clay liners, and the efficiency of these liners is impacted less by freeze-thaw cycles.

Geonet — A geonet is a plastic net-like drainage blanket that may be used in landfill liners in place of sand or gravel for the leachate collection layer. Sand and gravel are usually used due to cost considerations, and because geonets are more susceptible to clogging by small particles. This clogging would impair the performance of the LCS. Geonets do, however, convey liquid more rapidly than sand and gravel.

Figure 25[31,36] showed the schematic arrangement of a composite liner system and a simple configuration of GCLs. The thickness of geomembranes liner is of 0.060 in. for use in MSWLF. The design of the side slope, specifically the friction between natural soils and geosynthetics, is critical and requires careful review.



(a) Composite Liner System



(b) Configuration of Geosynthetic Clay Liners

Figure 25. Schematic Illustration About (a) Composite Liner System and (b) Configurations of GCLs.[31,36]

4.3.1.2.1. *Material types and thicknesses*

Geomembranes and or synthetic liners are made of one or more polymers along with a variety of other ingredients such as carbon black, pigments, fillers, plasticizers, processing aids, crosslinking chemicals, anti-degradants, and biocides. The polymers used to manufacture geomembranes include a wide range of plastics and rubbers differing in properties such as chemical resistance and basic composition. The polymeric materials may be categorized as follows:

- Thermoplastics such as polyvinyl chloride (PVC);
- Crystalline thermoplastics such as HDPE, very low-density polyethylene (VLDPE), and linear low-density polyethylene (LLDPE); and
- Thermoplastic elastomers such as chlorinated polyethylene (CPE) and chlorosulfonated polyethylene (CSPE).

The polymeric materials used most frequently as geomembranes are HDPE, PVC, CSPE, and CPE. The thicknesses of geomembranes range from 20 to 120 mil (1 mil = 0.001 in.). The recommended minimum thickness for all geomembranes is 30 mil, with the exception of HDPE, which must be at least 60 mil to allow for proper seam welding. Some geomembranes can be manufactured by a calendering process with fabric reinforcement, called scrim, to provide additional tensile strength and dimensional stability.

4.3.1.2.2. *Chemical and physical stress resistances*

The design of the landfill unit should consider stresses imposed on the liner by the design configuration. These stresses include the following:

- differential settlement in foundation soils;
- strain requirements at the anchor trench; and
- strain requirements over long and steep side slopes.

The chemical resistance of a geomembrane to leachate has traditionally been considered a critical issue for hazardous waste where highly concentrated solvents may be encountered. Chemical resistance testing of geomembranes may not be required for MSWLF units containing only MSW; due it that leachate from MSWLF units is not aggressive to these types of materials. Testing for chemical resistance may be warranted considering the waste type, volumes, characteristics, and amounts of small quantity generator waste or other industrial waste present in the waste stream. The following guidance is provided in the event such testing is of interest to the owner or operator.

US EPA's method 9090 in SW-846 is the established test procedure used to evaluate degradation of geomembranes when exposed to hazardous waste leachate.

In the procedure, the geomembrane is immersed in the site-specific chemical environment for at least 120 days at two different temperatures. Physical and mechanical properties of the tested material are then compared to those of the original material every 30 days. A software system, entitled flexible liner evaluation expert (FLEX), designed to assist in the hazardous waste permitting process. It is imperative that a geomembrane liner maintain its integrity during exposure to short- and long-term mechanical stresses. Short-term mechanical stresses include equipment traffic during the installation of a liner system, as well as thermal expansion and shrinkage of the geomembrane during the construction and operation of the MSWLF unit. Long-term mechanical stresses result from the placement of waste on top of the liner system and from subsequent differential settlement of the sub-grade. Long-term success of the liner requires adequate friction between the components of a liner system, particularly the soil subgrade and the geomembrane, and between geosynthetic components, so that slippage or sloughing does not occur on the slopes of the unit. Specifically, the foundation slopes and the subgrade materials must be considered in design equations to evaluate:

- The ability of a geomembrane to support its own weight on the side slopes;
- The ability of a geomembrane to withstand down-dragging during and after waste placement;
- The best anchorage configuration for the geomembrane;
- The stability of a soil cover on top of a geomembrane; and
- The stability of other geosynthetic components such as geotextile or geonet on top of a geomembrane.

These requirements may affect the choice of geomembrane material, including polymer type, fabric reinforcement, thickness, and texture (i.e. smooth or textured for HDPE). PVC also can be obtained in a roughened or file finish to increase the friction angle. Design specifications should indicate the type of raw polymer and manufactured sheet to be used as well as the requirements for the delivery, storage, installation, and sampling of the geomembrane. Material properties can be obtained from the manufacturer-supplied average physical property values, which are published in the Geotechnical Fabrics Report's Specifier's Guide and updated annually. The minimum tensile properties of the geomembrane must be sufficient to satisfy the stresses anticipated during the service life of the geomembrane. Specific raw polymer and manufactured sheet specifications and test procedures include:

Raw Polymer Specifications

- Density (ASTM D-1505);
- Melt index (ASTM D-1238);

- Carbon black (ASTM D-1603); and
- Thermogravimetric analysis (TGA) or differential scanning calorimetry (DSC).

Manufactured Sheet Specifications

- Thickness (ASTM D-1593);
- Tensile properties (ASTM D-638);
- Tear resistance (ASTM D-1004);
- Carbon black content (ASTM D- 1603);
- Carbon black dispersion (ASTM D-3015);
- Dimensional stability (ASTM D-1204); and
- Stress crack resistance (ASTM D- 1693).

Geomembranes may have different physical characteristics, depending on the type of polymer and the manufacturing process used, that can affect the design of a liner system. When reviewing manufacturers' literature, it is important to remember that each manufacturer may use more than one polymer or resin type for each grade of geomembrane and that the material specifications may be generalized to represent several grades of material.

4.3.1.2.3. *Installation*

Installation specifications should address installation procedures specific to the properties of the liner installed. The coefficient of thermal expansion of the geomembrane sheet can affect its installation and its service performance. The geomembrane should lie flat on the underlying soil. However, shrinkage and expansion of the sheeting, due to changes in temperature during installation, may result in excessive wrinkling or tension in the geomembrane. Wrinkles on the geomembrane surface will affect the uniformity of the soil-geomembrane interface and may result in leakage through imperfections. Excessive tautness of the geomembrane may affect its ability to resist rupture from localized stresses on the seams or at the toe of slopes where bridging over the sub-grade may occur during installation. In addition to thermal expansion and contraction of the geomembrane, residual stresses from manufacturing remain in some geomembranes and can cause nonuniform expansion and contraction during construction. Some flexibility is needed in the specifications for geomembrane selection to allow for anticipated dimensional changes resulting from thermal expansion and contraction.

Technical specifications for geomembrane also should include: information for protection of the material during shipping, storage, and handling; QC certifications provided by the manufacturer or fabricator (if panels are constructed); and QC testing by the contractor, installer, or a construction QA (CQA) agent. Installation procedures addressed by the technical specifications include a geomembrane layout plan, deployment of the geomembrane at the construction site, seam preparation, seaming methods, seaming temperature constraints, detailed procedures for repairing and documenting construction defects, and sealing of the geomembrane to appurtenances, both adjoining and penetrating the liner. The performance of inspection activities, including both nondestructive and destructive QC field testings of the sheets and seams during installation of the geomembrane, should be addressed in the technical specifications. The geomembrane sheeting is shipped in rolls or panels from the supplier, manufacturer, or fabricator to the construction site. Each roll or panel may be labeled according to its position on the geomembrane layout plan to facilitate installation. Upon delivery, the geomembrane sheeting should be inspected to check for damage that may have occurred during shipping. Proper storage of the rolls or panels prior to installation is essential to the final performance of the geomembrane. Some geomembrane materials are sensitive to ultraviolet exposure and should not be stored in direct sunlight prior to installation. Others, such as CSPE and CPE, are sensitive to moisture and heat and can partially crosslink or block (stick together) under improper storage conditions. Adhesives or welding materials, which are used to join geomembrane panels, also should be stored appropriately.

Visual inspection and acceptance of the soil liner sub-grade should be conducted prior to installing the geomembrane. The surface of the sub-grade should meet design specifications with regard to lack of protruding objects, grades, and thickness. Once these inspections are conducted and complete, the geomembrane may be installed on top of the soil liner. If necessary, other means should be employed to protect the sub-grade from precipitation and erosion, and to prevent desiccation, moisture loss, and erosion from the soil liner prior to geomembrane placement. Such methods may include placing a plastic tarp on top of completed portions of the soil liner. In addition, scheduling soil liner construction slightly ahead of the geomembrane and drainage layer placement can reduce the exposure of the soil liner to the elements.

Deployment or placement of the geomembrane panels or rolls should be described in the geomembrane layout plan. Rolls of sheeting, such as HDPE, generally can be deployed by placing a shaft through the core of the roll, which is supported and deployed using a front-end loader or a winch. Panels composed of extremely flexible liner material such as PVC are usually folded on pallets, requiring workers to manually unfold and place the geomembrane. Placement of the geomembrane goes hand-in-hand with the seaming process; no more than the amount of sheeting that can be seamed during a shift or work day should be deployed at any one time. Panels should be weighted with sand bags if wind uplift of the membrane or excessive movement from thermal expansion is a potential problem. Proper stormwater control measurements should be employed during construction to prevent erosion of the soil liner underneath the geomembrane and the washing away of the geomembrane. Once deployment of a section of the geomembrane is complete and each section has been visually inspected for imperfections and tested to ensure that it is the specified thickness, seaming of the geomembrane may begin. QC/QA monitoring of the seaming process should be implemented to detect inferior seams. Seaming can be conducted either in the factory or in the field. Factory seams are made in a controlled environment and are generally of high quality, but the entire seam length (100%) still should be tested nondestructively.

Consistent quality in fabricating field seams is critical to liner performance, and conditions that may affect seaming should be monitored and controlled during installation. An inspection should be conducted in accordance with a CQA plan to document the integrity of field seams. Factors affecting the seaming process include:

- Ambient temperature at which the seams are made;
- Relative humidity;
- Control of panel lift-up by wind;
- The effect of clouds on the geomembrane temperature;
- Water content of the subsurface beneath the geomembrane;
- The supporting surface on which the seaming is bonded;
- The skill of the seaming crew;
- Quality and consistency of the chemical or welding material;
- Proper preparation of the liner surfaces to be joined;
- Moisture on the seam interface; and
- Cleanliness of the seam interface (i.e. the amount of airborne dust and debris present).

Depending on the type of geomembrane, several bonding systems are available for the construction of both factory and field seams. Bonding methods include solvents, heat seals, heat guns, dielectric seaming, extrusion welding, and hot wedge techniques. To ensure the integrity of the seams, a geomembrane should be seamed using the bonding system recommended by the manufacturer. Thermal methods of seaming require cleanliness of the bonding surfaces, heat, pressure, and dwell time to produce high-quality seams. The requirements for adhesive systems are the same as those for thermal systems, except that the adhesive takes the place of the heat. Sealing the geomembrane to appurtenances and penetrating structures should be performed in accordance with detailed drawings included in the design plans and approved specifications. An anchor trench along the perimeter of the cell generally is used to secure the geomembrane during construction (to prevent sloughing or slipping down the interior side slopes). Run out calculations are available to determine the depth of burial at a trench necessary to hold a specified length of membrane, or combination of membrane and geofabric or geotextile. If forces larger than the tensile strength of the membrane are inadvertently developed, then the membrane could tear. For this reason, the geomembrane should be allowed to slip or give in the trench after construction to prevent such tearing. However, during construction, the geomembrane should be anchored according to the detailed drawings provided in the design plans and specifications. Geomembranes that are subject to damage from exposure to weather and work activities should be covered with a layer of soil as soon as possible after QA activities associated with geomembrane testing are completed. Soil should be placed without driving construction vehicles directly on the geomembrane. Light ground pressure bulldozers may be used to push material out in front over the liner, but the operator must not attempt to push a large pile of soil forward in a continuous manner over the membrane. Such methods can cause localized wrinkles to develop and overturn in the direction of movement. Overturned wrinkles create sharp creases and localized stresses in the geomembrane that could lead to premature failure. Instead, the operator should continually place smaller amounts of soil or drainage material working outward over the toe of the previously placed material. Alternatively, large backhoes can be used to place soil over the geomembrane that can later be spread with a bulldozer or similar equipment. Although such methods may sound tedious and slow; in the long run, they will be faster and more cost-effective than placing too much material too fast and having to remobilize the liner installer to repair damaged sections of the geomembrane. The QA activities conducted during construction also should include monitoring the contractor's activities on top of the liner to avoid damage to installed and accepted geomembranes.

4.3.2. *Leachate Collection Systems*

Leachate refers to liquid that has passed through or emerged from solid waste and contains dissolved, suspended, or immiscible materials removed from the solid waste. At MSWLF units, leachate is typically aqueous with limited, if any, immiscible fluids or dissolved solvents. The primary function of the LCS is to collect and convey leachate out of the landfill unit and to control the depth of the leachate above the liner. The LCS should be designed to meet the concern authorities regulatory performance standard of maintaining <30 cm (12 in.) depth of leachate, or "head," above the liner. The 30-cm head allowance is a normal design standard and this design standard may be exceeded for relatively short periods during the active life of the MSWLF unit. Flow of leachate through imperfections in the liner system increases with an increase in leachate head above the liner. Maintaining a low-leachate level above the liner helps to improve the performance of the composite liner.

Leachate is generally collected from the landfill through sand drainage layers, synthetic drainage nets, or granular drainage layers with perforated plastic collection pipes, and is then removed through sumps or gravity drain carrier pipes. LCS's should consist of the following components:

- A low-permeability base (in this case a composite liner);
- A high-permeability drainage layer, constructed of either natural granular materials (sand and gravel) or synthetic drainage material (i.e. geonet) placed directly on the FML, or on a protective bedding layer (i.e. geofabric) directly overlying the liner;
- Perforated leachate collection pipes within the high-permeability drainage layer to collect leachate and carry it rapidly to a sump or collection header pipe;
- A protective filter layer over the high-permeability drainage material, if necessary, to prevent physical clogging of the material by fine-grained material; and
- Leachate collection sumps or header pipe system where leachate can be removed.

The design, construction, and operation of the LCS should maintain a maximum height of leachate above the composite liner of 30 cm (12 in.). Design guidance for calculating the maximum leachate depth over a liner for granular drainage systems materials is provided US EPA documents.[29–32] The leachate head in the layer is a function of the liquid impingement rate, bottom slope, pipe spacing, and drainage layer hydraulic conductivity. The impingement rate is estimated using a complex liquid routing procedure. If the maximum leachate depth exceeds 30 cm for the system, except for short-term occurrences, the design should be modified to improve its efficiency by increasing grade, decreasing pipe spacing, or increasing the hydraulic conductivity (transmissivity) of the drainage layer.

4.3.2.1. Grading of low-permeability base

The typical bottom liner slope is a minimum of 2% after allowances for settlement at all points in each system. A slope is necessary for effective gravity drainage through the entire operating and post-closure period. Settlement estimates of the foundation soils should set this 2% grade as a post-settlement design objective.

4.3.2.2. High-permeability drainage layer

The high-permeability drainage layer is placed directly over the liner or its protective bedding layer at a slope of at least 2% (the same slope necessary for the composite liner). Often the selection of a drainage material is based on the on-site availability of natural granular materials. In some regions of the country, hauling costs may be very high for sand and gravel, or appropriate materials may be unavailable; therefore, the designer may elect to use geosynthetic drainage nets (geonets) or synthetic drainage materials as an alternative. Frequently, geonets are substituted for granular materials on steep sidewalls because maintaining sand on the slope during construction and operation of the landfill unit is more difficult.

4.3.2.3. Soil drainage layers

If the drainage layer of the LCS is constructed of granular soil materials (i.e. sand and gravel layer in Fig. 25), then it should be demonstrated that this granular drainage layer has sufficient bearing strength to support expected loads. This demonstration will be similar to that required for the foundations and soil liner. If the landfill unit is designed on moderate-to-steep (15%) grades, the landfill design should include calculations demonstrating that the selected granular drainage materials will be stable on the most critical slopes (i.e. usually the steepest slope) in the design. The calculations and assumptions should be shown, especially the friction angle between the geomembrane and soil, and if possible, supported by laboratory and/or field testing. Generally, gravel soil can be expected to have a hydraulic conductivity of greater than 0.01 cm/s, while sands can be expected to have a coefficient of permeability greater than 0.001 cm/s. The sand or gravel drains leachate that enters the drainage layer to prevent 30 cm (12 in.) or more accumulation on top of the liner during the active life of the MSWLF unit LCS. The design of an LCS frequently uses a drainage material with a hydraulic conductivity of 1×10^{-2} cm/s or higher. Drainage materials with hydraulic conductivities in this order of magnitude should be evaluated for biological and particulate clogging. Alternatively, if a geonet is used, the design is based on the transmissivity of the geonet. If a filter layer (soil or geosynthetic) is constructed on top of a drainage layer to protect it from clogging, and the LCS is designed and operated to avoid drastic changes in the oxidation reduction potential of the leachate (thereby avoiding formation of precipitates within the LCS), then there is no conceptual basis to anticipate that conductivity will decrease over time. Where conductivity is expected to decrease over time, the change in impingement rate also should be evaluated over the same time because the reduced impingement rate and hydraulic conductivity may still comply with the 30 cm criterion. Unless alternative provisions are made to control incident precipitation and resulting surface run-off, the impingement rate during the operating period of the MSWLF unit is usually at least an order of magnitude greater than the impingement rate after final closure. The critical design condition for meeting the 30 cm (12 in.) criterion can, therefore, be expected during the operating life. The designer may evaluate the sensitivity of a design to meet the 30 cm (12 in.) criterion because of changes in impingement rates, hydraulic conductivity, pipe spacing, and grades. Such sensitivity analysis may indicate which element of the design should be emphasized during construction quality monitoring or whether the design can be altered to comply with the 30 cm (12 in.) criterion in a more cost-effective manner.

The soil material used for the drainage layer should be investigated at the borrow pit prior to use at the landfill. Typical borrow pit characterization testing would include laboratory hydraulic conductivity and grain size distribution. If grain size distribution information from the borrow pit characterization program can be correlated to the hydraulic conductivity data, then the grain size test, which can be conducted in a short time in the field, may be a useful construction QC parameter. Compliance with this parameter would then be indicative that the hydraulic conductivity design criterion was achieved in the constructed drainage layer. This information could be incorporated into construction documents after the borrow pit has been characterized. If a correlation cannot be made between hydraulic conductivity and grain size distribution, then construction documents may rely on direct field or laboratory measurements to demonstrate that the hydraulic conductivity design criterion was met in the drainage layer.

Granular materials are generally placed using conventional earthmoving equipment, including trucks, scrapers, bulldozers, and front-end loaders. Vehicles should not be driven directly over the geosynthetic membrane when it is being covered. Coarse granular drainage materials, unlike low-permeability soils, can be placed dry and do not need to be heavily compacted. Compacting granular soils tends to grind the soil particles together, which increases the fine material and reduces hydraulic conductivity. To minimize settlement following material placement, the granular material may be compacted with a vibratory roller. The final thickness of the drainage layer should be checked by optical survey measurements or by direct test pit measurements.

4.3.2.4. Geosynthetic drainage nets

Geosynthetic drainage nets (geonets) may be substituted for the granular layers of the LCRs on the bottom and sidewalls of the landfill cells. Geonets require less space than perforated pipe or gravel and also promote rapid transmission of liquids. They do, however, require geotextile filters above them and can experience problems with creep and intrusion. Long-term operating and performance experience of geonets is limited because the material and its application are relatively new. If a geonet is used in place of a granular drainage layer, it must provide the same level of performance (maintaining $\leq$30 cm of leachate head above the liner). The transmissivity of a geonet can be reduced significantly by intrusion of the soil or a geotextile. A protective geotextile between the soil and geonet will help alleviate this concern. If laboratory transmissivity tests are performed, they should be done under conditions, loads, and configurations that closely replicate the actual field conditions. It is important that the transmissivity value used in the LCS design calculations be selected based upon those loaded conditions. It is also important to ensure that appropriate factors of

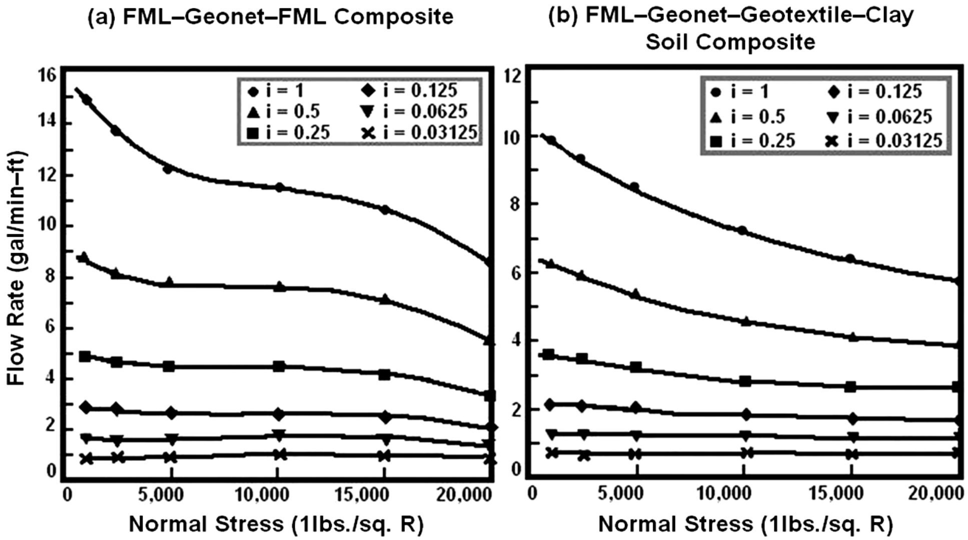


Figure 26. Flow Rate Curves for Geonets in Two Composite Liner Configurations.[29]

safety are used. The flow rate or transmissivity of geonets may be evaluated by ASTM D-4716. This flow rate may then be compared to design-by-function equations. In the ASTM D-4716 flow test, the proposed collector cross section should be modeled as closely as possible to actual field conditions.

Figure 26[29] showed the flow rate "signatures" of a geonet between two geomembranes [*left curves (a)*] and the same geonet between a layer of clay soil and a geomembrane [*right curves (b)*]. The differences between the two sets of curves represent intrusion of the geotextile/clay into the apertures of the geonet.

Generally, geonets perform well and result in high factors of safety or performance design ratios, unless creep (elongation under constant stress) becomes a problem or adjacent materials intrude into apertures. For geonets, the most critical specification is the ability to transmit fluids under load. The specifications also should include a minimum transmissivity under expected landfill operating (dynamic) or completion (static) loads. The specifications for thickness and types of material should be identified on the drawings or in the materials section of the specifications, and should be consistent with the design calculations. Geonets are often used on the sidewalls of landfills because of their ease of installation. They should be placed with the top ends in a secure anchor trench with the strongest longitudinal length extending down the slope. The geonets need not be seamed to each other on the slopes, only tied at the edges, butted, or overlapped. They should be placed in a loose condition, not stretched or placed in a configuration where they are bearing their own weight in tension. The construction specifications should contain appropriate installation requirements as described above or the requirements of the geonet manufacturer. All geonets need to be protected by a filter layer or geotextile to prevent clogging. The friction factors against sliding for geotextiles, geonets, and geomembranes often can be estimated using manufacturers data because these materials do not exhibit the range of characteristics as seen in soil materials. However, it is important that the designer performs the actual tests using site materials and that the sliding stability calculations accurately represent the actual design configuration, site conditions, and the specified material characteristics.

4.3.3. *Leachate Collection Pipes*

All components of the LCS must have sufficient strength to support the weight of the overlying waste, cover system, and post-closure loadings, as well as the stresses from operating equipment. The component that is most vulnerable to compressive strength failure is the drainage layer piping. LCS piping can fail by excessive deflection, which may lead to buckling or collapse. Pipe strength calculations should include resistance to wall crushing, pipe deflection, and critical buckling pressure. Design equations and information for most pipe types can be obtained from the major pipe manufacturers. Perforated drainage pipes can provide good long-term performance. These pipes have been shown to transmit fluids rapidly and to maintain good service lives. The depth of the drainage layer around the pipe should be deeper than the diameter of the pipe. The pipes can be placed in trenches to provide the extra depth. In addition, the trench serves as a sump (low point) for leachate collection. Pipes can be susceptible to particulate and biological clogging similar to the drainage layer material. Furthermore, pipes also can be susceptible to deflection. Proper maintenance and design of pipe systems can mitigate these effects and provide systems that function properly. The design of perforated collection pipes should consider the following factors:

- The required flow using known percolation impingement rates and pipe spacing;
- Pipe size using required flow and maximum slope; and
- The structural strength of the pipe.

The pipe spacing may be determined by the *mound model* that is illustrated in Fig. 27.[29] In the *mound model*, the maximum height of liquid between two parallel perforated drainage pipes is equal to:

$$h_{\max} = \frac{L\sqrt{c}}{2}\left[\frac{\tan^2\alpha}{c} + 1 - \frac{\tan\alpha}{c}\sqrt{\tan^2\alpha + c}\right] \quad (2)$$

where $c = q/k$, k = permeability, q = inflow rate, and α = slope.

However, two unknown in the equation are L = distance between the pipes and C = amount of leachate.

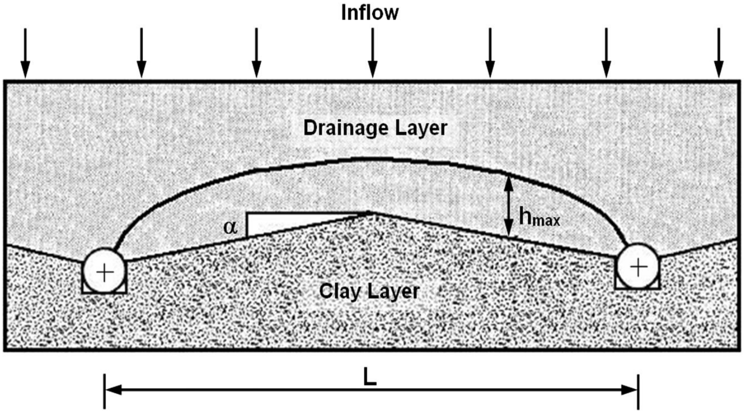


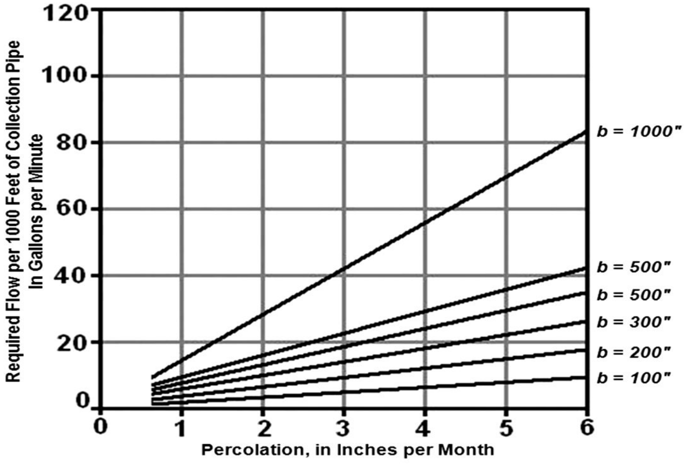
Figure 27. Definition of Terms for Mound Model Flow Rate Calculations.[29]

Using a maximum allowable head, $h_{\max}$ of 30 cm (12 in.), the equation is usually solved for "L" which showed in Fig. 27.

The amount of leachate, "c," can be estimated in a variety of ways including the "water balance method" and the computer, i.e. HELP. The HELP Model is a quasi-2D hydrologic model of water movement across, into, through, and out of landfills. The model uses climatologic, soil, and landfill design data and incorporates a solution technique that accounts for the effects of surface storage, run-off, infiltration, percolation, soil-moisture storage, evapotranspiration, and lateral drainage. The program estimates run-off drainage and leachate that are expected to result from a wide variety of landfill conditions, including open, partially open, and closed landfill cells. The model also may be used to estimate the depth of leachate above the bottom liner of the landfill unit. The results may be used to compare designs or to aid in the design of LCSs.

Once the percolation and pipe spacing are known, the design flow rate can be obtained using the curve in Fig. 28.[29] The amount of leachate percolation at the particular site is located on the x-axis.

The required flow rate is the point at which this value intersects with the pipe spacing value determined from the *mound model* (Fig. 27). Using this value of flow rate and the bottom slope of the site, the required diameter for the pipe can be determined from Fig. 29.[29] Finally, the illustrated graphics in Figs. 30[29] and 31[29] showed two ways to determine whether or not the strength of the pipe is adequate for the landfill design. In Fig. 30, the vertical soil pressure is located on the y-axis. The density of the backfill material around the pipe is not governed by strength; therefore, it will deform under pressure rather than break. As much as 10% is the absolute limiting deflection value for plastic pipe. By using Fig. 30, the applied



b = width of area contributing to Leachate collection pipe

Figure 28. Required Capacity of Leachate Collection Pipe.[29]

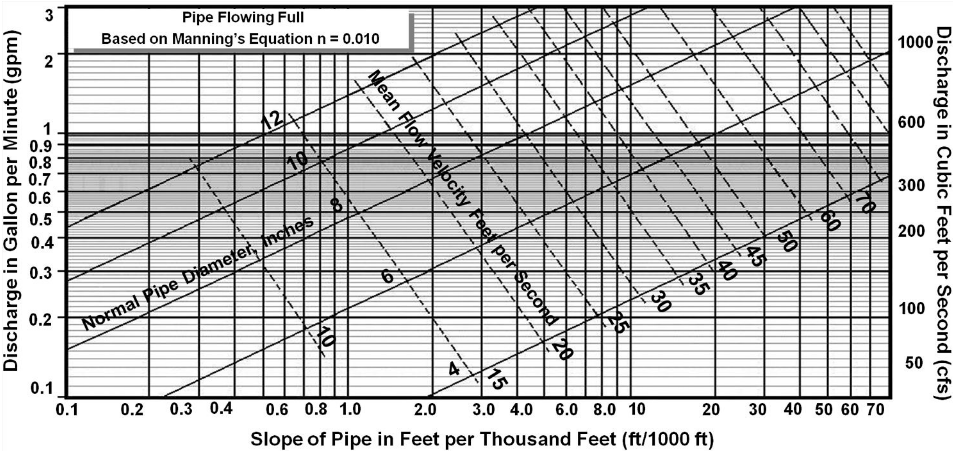


Figure 29. Leachate Collection Pipe Sizing Chart.[29]

pressure on the pipe is located and traced to the trench geometry, and then the pipe deflection value is checked for its adequacy.

The LCS specifications should include:

- type of piping material;
- diameter and wall thickness;

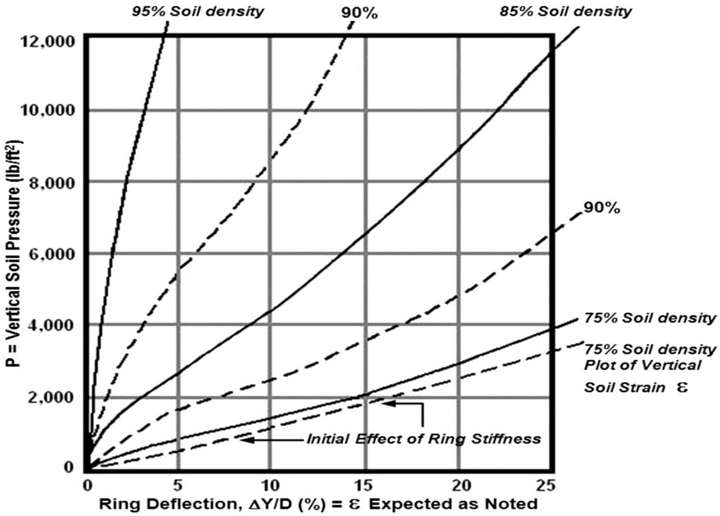


Figure 30. Vertical Ring Deflection Versus Vertical Soil Pressure for 18-in. Corrugated Polyethylene in High-pressure Soil Cell.[29]

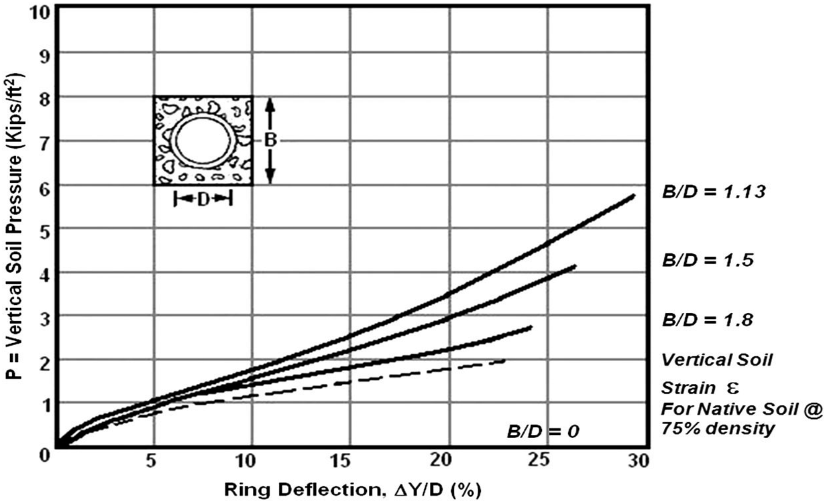


Figure 31. Effect of Trench Geometry and Pipe Sizing on Ring Deflection.[29]

- size and distribution of slots and perforations;
- type of coatings (if any) used in the pipe manufacturing; and
- type of pipe bedding material and required compaction used to support the pipes.

The construction drawings and specifications should clearly indicate the type of bedding to be used under the pipes and the dimensions of any trenches. The specifications should indicate how the pipe lengths are joined. The drawings should show how the pipes are placed with respect to the perforations. To maintain the lowest possible leachate head, there should be perforations near the pipe invert, but not directly at the invert. The pipe invert itself should be solid to allow for efficient pipe flow at low volumes. When drainage pipe systems are embedded in filter and drainage layers, no unplugged ends should be allowed. The filter materials in contact with the pipes should be appropriately sized to prevent migration of the material into the pipe. The filter media, drainage layer, and pipe network should be compatible and should represent an integrated design.

4.3.3.1. Protection of leachate collection pipes

The long-term performance of the LCS depends on the design used to protect pipes from physical clogging (sedimentation) by the granular drainage materials. Use of a graded material around the pipes is most effective if accompanied by proper sizing of pipe perforations. Design criteria have established by using graded filters to prevent physical clogging of leachate drainage layers and piping by soil sediment deposits. When installing graded filters, caution should be taken to prevent segregation of the material. Clogging of the pipes and drainage layers of the LCS can occur through several other mechanisms, including chemical and biological foulings. The LCS should be designed with a cleanout access capable of reaching all parts of the collection system with standard pipe-cleaning equipment.

Chemical clogging can occur when dissolved species in the leachate precipitate in the piping. Clogging can be minimized by periodically flushing pipes or by providing a sufficiently steep slope in the system to allow for high flow velocities for self-cleansing. These velocities are dependent on the diameter of the precipitate particles and on their specific gravity. Generally, flow velocities should be in the range of 1–2 ft/s to allow for self-cleansing of the piping.

Biological clogging due to algal and bacterial growths can be a serious problem in MSWLF units. There are no universally effective methods of preventing such biological growths. Since organic materials will be present in the landfill unit, there will be a potential for biological clogging. The system design should include features that allow for pipe system cleanings. The components of the cleaning system should include:

- A minimum of 6-in. diameter pipes to facilitate cleaning;
- Access located at major pipe intersections or bends to allow for inspections and cleaning; and
- Valves, ports, or other appurtenances to introduce biocides and/or cleaning solutions.

4.3.3.2. Protection of the high-permeability drainage layer

The openings in drainage materials, whether holes in pipes, voids in gravel, or apertures in geonets, must be protected against clogging by accumulation of fine (silt-sized) materials. An intermediate material that has smaller openings than those of the drainage material can be used as a filter between the waste and drainage layer. Sand may be used as filter material, but has the disadvantage of taking up vertical space. Geotextiles do not use up air space and can be used as filter materials.

4.3.3.3. Soil filter layers

There are three parts to an analysis of a sand filter that is placed above drainage material. The first determines whether or not the filter allows adequate flow of liquids. The second evaluates whether or not the void spaces are small enough to prevent solids from being lost from the upstream materials. The third estimates the long-term clogging behavior of the filter. The particle-size distribution of the drainage system and the particle-size distribution of the invading (or upstream) soils are required in the design of granular soil (sand filter) materials. The filter material should have its large- and small-sized particles intermediate between the two extremes. Equations for adequate flow and retention are:

- Adequate Flow:

$$d_{85\mathrm{f}} > (3{-}5)d_{15\mathrm{d.s.}} \tag{3}$$

- Adequate Retention:

$$d_{15\mathrm{f}} < (3{-}5)d_{85\mathrm{w.f.}} \tag{4}$$

where f = required filter soil; d.s. = drainage stone; and w.f. = water fines.

There are no quantitative methods to assess soil filter clogging, although empirical guidelines are found in geotechnical engineering references.

The specifications for granular filter layers that surround perforated pipes and that protect the drainage layer from clogging are based on a well-defined particle size distribution. The orientation and configuration of filter layers relative to other LCS components should be shown on all drawings and should be described, with ranges of particle sizes. Thickness is an important placement criterion for granular filter material. Generally, the granular filter materials will be placed around perforated pipes by hand, forming an "envelope." The dimensions of the envelope should be clearly stated on the drawings or in the specifications. This envelope can be placed at the same time as the granular drainage layer, but it is important that the filter envelope protect all areas of the pipe where the clogging potential exists. The plans and specifications should indicate the extent of the envelope. The construction QC program should document that the envelope was installed according to the plans and specifications. A granular filter layer is generally placed using the same earthmoving equipment as the granular drainage layer. The final thickness should be checked by optical survey or by direct test pit measurement. This filter layer is the uppermost layer in the LCS. A landfill design option includes a buffer layer, 12-in. thick (30 cm) or more, to protect the filter layer and drainage layer from damage due to traffic. This final layer can be general fill, as long as it is no finer than the soil used in the filter layer. However, if the layer has a low permeability, it will affect leachate recirculation attempts.

4.3.3.4. Geotextile filter layers

Geotextile filter fabrics allow liquid flow while simultaneously preventing upstream fine particles from fouling the drain. Geotextiles save vertical space, are easy to install, and have the added advantage of remaining stationary under load. Geotextiles also can be used as cushioning materials above geomembranes due to. These filters are susceptible to biological clogging, their use in areas inundated by leachate (i.e. sumps, around leachate collection pipes, and trenches) should be avoided. Geotextile filter design parallels sand filter design with some modifications. Adequate flow is assessed by comparing the material (allowable) permittivity to the design imposed permittivity. The design permittivity utilizes an adapted form of *Darcy's law*. The resulting comparison yields a design ratio, or FS, that is the focus of the design:

$$\mathrm{DR} = \text{Ø}_{\mathrm{allow}}/\text{Ø}_{\mathrm{reqd}} \tag{5}$$

where $\text{Ø}_{\mathrm{allow}}$ = permittivity from ASTM D-4491; $\text{Ø}_{\mathrm{reqd}} = (q/a)(1/h_{\max})$; $q/a =$ inflow rate per unit area; and $h_{\max} =$ 12-in. or 30 cm.

The second part of the geotextile filter design is determining the opening size necessary for retaining the upstream soil or particulates in the leachate. It is well established that the 95% opening size is related to particles to be retained in the following type of relationship:

$$O_{95} < \mathrm{fct.}(d_{50}, \mathrm{CU}, \mathrm{DR}), \tag{6}$$

where O_{95} = 95% opening size of geotextile; d_{50} = 50% size of upstream particles; CU = uniformity of the upstream particle size; and DR = relative density of the upstream particles.

The O_{95} size of a geotextile in the equation is the opening size at which 5% of a given value should be less than the particle size characteristics of the invading materials. In the test for the O_{95} size of the geotextile, a sieve with a very coarse mesh in the bottom is used as a support. The geotextile is placed on top of the mesh and is bonded to the inside, so that the glass beads used in the test cannot escape around the edges of the geotextile filter. The particle-size distribution of retained glass beads is compared to the allowable value using any of a number of existing formulas.

The third consideration in geotextile design is long-term clogging. A test method for this problem that may be adopted by ASTM is called the "gradient ratio test." In this test, the hydraulic gradient of 1 in. of soil plus the underlying geotextile is compared with the hydraulic gradient of 2 in. of soil. The higher the gradient ratio is, the more likely that a clog will occur. The final ASTM gradient ratio test will include failure criteria. An alternative to this test method is a long-term flow test that also is performed in a laboratory. The test models a soil-to-fabric system at the anticipated hydraulic gradient. The flow rate through the system is monitored. A long-term flow rate will gradually decrease until it stops altogether.

The primary function of a geotextile is to prevent the migration of fines into the leachate pipes while allowing the passage of leachate. The most important specifications are those for hydraulic conductivity and retention. The hydraulic conductivity of the geotextile generally should be at least 10 times the soil it is retaining. An evaluation of the retention ability for loose soils is based on the average particle size of the soil and the apparent opening size (AOS) of the geotextile. The maximum AOS, sometimes called equivalent opening size, is determined by the size of the soil that will be retained; a geotextile is then selected to meet that specification. The material specifications should contain a range of AOS values for the geotextile, and these AOS values should match those used in the design calculations.

One of the advantages of geotextiles is their light-weight and ease of placement. The geotextiles are brought to the site, unrolled, and held down with sandbags until they are covered with a protective layer. They are usually overlapped, not seamed; however, on slopes or in other configurations, they may be sewn. As with granular filter layers, it is important that the design drawings be clear in their designation of geotextile placement, so that no potential route of pipe or drainage layer clogging is left unprotected. If geotextiles are used on a slope, they should be secured in an anchor trench similar to those for geomembranes or geonets.

4.3.4. *CQA and QC*

CQA consists of a planned series of observations and tests to ensure that the final product meets project specifications. CQA plans, specifications, observations, and tests are used to provide quantitative criteria with which to accept the final product. On routine construction projects, CQA is normally the concern of the owner and is obtained using an independent third-party testing firm. The independence of the third-party inspection firm is important, particularly when the owner is a corporation or other legal entity that has under its corporate "umbrella" the capacity to perform the CQA activities. Although "in-house" CQA personnel may be registered professional engineers, a perception of misrepresentation may exist if CQA is not performed by an independent third party.

CQC is an on-going process of measuring and controlling the characteristics of the product in order to meet manufacturer's or project specifications. CQC is a production tool that is employed by the manufacturer of materials and by the contractor installing the materials at the site. CQA, by contrast, is a verification tool employed by the facility owner or regulatory agency to ensure that the materials and installations meet project specifications. CQC is performed independently of the CQA plan. For example, while a geomembrane liner installer will perform CQC testing of field seams, the CQA program will require independent CQA testing of those same seams by a third-party inspector.

The CQA/CQC plans are implemented through inspection activities that include visual observations, field testing and measurements, laboratory testing, and evaluation of the test data. Inspection activities typically are concerned with four separate functions:

1. *QC inspection by the manufacturer* provides an in-process measure of the product quality and its conformance with the project plans and specifications. Typically, the manufacturers will QC test results to certify that the product conforms to project plans and specifications.
2. *CQC inspection by the contractor* provides an in-process measure of construction quality and conformance with the project plans and specifications, thereby allowing the contractor to correct the construction process if the quality of the product is not meeting the specifications and plans.
3. *CQA testing by the owner* (i.e. acceptance inspection) performed by the owner usually through the third-party testing firm, provides a measure of the final product quality and its conformance with project plans and specifications. Due to the size and costs of a typical MSWLF unit construction project, rejection of the project at completion would be costly to all parties. Acceptance inspections as portions of the project become complete allow deficiencies to be found and corrected before they become too large and costly.
4. *Regulatory inspection* often is performed by a regulatory agency to ensure that the final product conforms with all applicable codes and regulations. In some cases, the regulatory agency will use CQA documentation and the as-built plans or "record drawings" to confirm compliance with the regulations.

4.3.4.1. Soil liner QA/QC

QC testing performed on materials used in construction of the landfill unit includes source testing and construction testing. Source testing defines material properties that govern material placement. Source testing commonly includes moisture content, soil density, Atterberg limits, grain size, and laboratory hydraulic conductivity. Construction testing ensures that landfill construction has been performed in accordance with the plans and technical specifications. Construction testing generally includes tests of soil moisture content, density, lift thickness, and hydraulic conductivity.

The method of determining compliance with the maximum hydraulic conductivity criterion should be specified in the QA/QC plan. Some methods have included the use of the criterion as a maximum value that never should be exceeded, while other methods have used statistical techniques to estimate the true mean. The sample collection program should be designed to work with the method of compliance determination. Selection of sample collection points should be made on a random basis. Thin-walled sampling tubes generally are used to collect compacted clay samples for laboratory hydraulic conductivity testing. It is important to minimize disturbance of the sample being collected. Tubes pushed into the soil by a backhoe may yield disturbed samples. A recommended procedure (when a backhoe is available during sample collection) is to use the backhoe bucket as a stationary support and push the tube into the clay with a jack positioned between the clay and the tube. The sample hole should be filled with bentonite or a bentonite clay mixture, and compacted using short lifts of material. If geophysical methods are used for moisture and density measurements, it is recommended that alternative methods be used less frequently to verify the accuracy of the faster geophysical methods.

QA testing for soil liners includes the same testing requirements as specified above for control testing. Generally, the tests are performed less frequently and are performed by an individual or an entity independent of the contractor. Activities of the CQA are essential to document quality of construction. The CQA official's responsibilities may include:

- Communicating with the contractor;
- Interpreting and clarifying project drawings and specifications with the designer, owner, and contractor;
- Recommending acceptance or rejection by the owner/operator of work completed by the construction contractor;
- Submitting blind samples (e.g., duplicates and blanks) for analysis by the contractor's testing staff or one or more independent laboratories, as applicable;
- Notifying owner or operator of construction quality problems not resolved on-site in a timely manner;

- Observing the testing equipment, personnel, and procedures used by the construction contractor to check for detrimentally significant changes over time;
- Reviewing the construction contractor's QC recording, maintenance, summary, and interpretations of test data for accuracy and appropriateness; and
- Reporting to the owner/operator on monitoring results.

4.3.4.2. Soil liner pilot construction

A pilot construction or test fill is a small-scale test pad that can be used to verify that the soil, equipment, and construction procedures can produce a liner that performs according to the construction drawings and specifications. An owner or operator may want to consider the option of constructing a test fill prior to the construction of the liner. A test pad is useful not only in teaching people how to build a soil liner, it also can function as a CQA tool. If the variables used to build a test pad that achieves a 1×10^{-7} cm/s hydraulic conductivity are followed exactly, then the completed full-size liner should meet the regulatory requirements. A test fill may be a cost-effective method for the contractor to evaluate the construction methods and borrow source. Specific factors that can be examined/tested during construction of a test fill include:

- Preparation and compaction of foundation material to the required bearing strength;
- Methods of controlling uniformity of the soil material;
- Compactive effort (e.g., type of equipment, number of passes) to achieve required soil density and hydraulic conductivity;
- Lift thickness and placement procedures to achieve uniformity of density throughout a lift and the absence of apparent boundary effects between lifts or between placements in the same lift;
- Procedures for protecting against desiccation cracking or other site- and season-specific failure mechanisms for the finished liner or intermediate lifts;
- Measuring the hydraulic conductivity on the test fill in the field and collecting samples of field-compacted soil for laboratory testing;
- Test procedures for controlling the quality of construction;
- Ability of different types of soil to meet hydraulic conductivity requirements in the field; and
- Skill and competence of the construction team, including equipment operators and QC specialists.

4.3.4.3. Geomembrane QA/QC testing

As with the construction of soil liners, installation of geomembrane liners should be in conformance with a QA/QC plan. Tests performed to evaluate the integrity of geomembrane seams are generally considered to be either "destructive" or "nondestructive."

4.3.4.3.1. *Destructive testing*

QC testing of geomembranes generally includes peel and shear testings of scrap test weld sections prior to commencing seaming activities and at periodic intervals throughout the day. Additionally, destructive peel and shear field tests are performed on samples from the installed seams. QA testing generally requires that an independent laboratory perform peel and shear tests of samples from installed seams. The samples may be collected randomly or in areas of suspect quality. HDPE seams are generally tested at intervals equivalent to one sample per every 300–400 ft of installed seam for extrusion welds, and every 500 ft for fusion-welded seams. Extrusion seams on HDPE require grinding prior to welding, which can greatly diminish parent material strengths if excessive grinding occurs. For dual hot wedge seams in HDPE, both the inner and outer seams may be subjected to destructive shear tests at the independent laboratory. Destructive samples of installed seam welds are generally cut into several pieces and distributed to:

- The installer to perform construction QC field testing;
- The owner/operator to retain and appropriately catalog or archive; and
- An independent laboratory for peel and shear testing.

If the test results for a seam sample do not pass the acceptance/rejection criteria, then samples are cut from the same field seam on both sides of the rejected sample location. Samples are collected and tested until the area limits of the low-quality seam are defined. Corrective measures should be undertaken to repair the length of seam that has not passed the acceptance/rejection criteria. In many cases, this involves seaming a cap over the length of the rejected seam or reseaming the affected area. In situations where the seams continually fail testing, the seaming crews may have to be retrained.

4.3.4.3.2. *Nondestructive testing*

Nondestructive test methods are conducted in the field on an in-place geomembrane. These test methods determine the integrity of the geomembrane field seams. Nondestructive test methods include the probe test, air lance, vacuum box, ultrasonic methods (pulse echo, shadow, and impedance planes), electrical spark test, pressurized dual seam, electrical resistivity, and hydrostatic tests. Seam sections that fail appropriate, nondestructive tests must be carefully delineated, patched or reseamed, and retested. Large patches or reseamed areas should be subjected to destructive test procedures for QA purposes. The specifications should clearly describe the degree to which nondestructive and destructive test methods will be used in evaluating failed portions of nondestructive seam tests.

4.3.4.4. Geomembrane CQA activities

The responsibilities of the CQA personnel for the installation of the geomembrane are generally the same as the responsibilities for the construction of a soil liner with the following additions:

- Observation of liner storage area and liners in storage, and handling of the liner as the panels are positioned in the cell;
- Observation of seam overlap, seam preparation prior to seaming, and material underlying the liner;
- Observation of destructive testing conducted on scrap test welds prior to seaming;
- Observation of destructive seam sampling, submission of the samples to an independent testing laboratory, and review of results for conformance to specifications;
- Observation of all seams and panels for defects due to manufacturing and/or handling and placement;
- Observation of all pipe penetration boots and welds in the liner;
- Preparation of reports indicating sampling conducted and sampling results, locations of destructive samples, locations of patches, locations of seams constructed, and any problems encountered; and
- Preparation of record drawings of the liner installation, in some cases.

4.3.4.5. LCS CQA

The purpose of LCS CQA is to document that the system construction is in accordance with the design specifications. Prior to construction, all materials should be inspected to confirm that they meet the construction plans and specifications. These include:

- geonets;
- geotextiles;
- geosynthetics;
- pipe size, materials, and perforations;
- granular material gradation and prefabricated structures (sumps, manholes, etc.);
- mechanical, electrical, and monitoring equipments; and
- concrete forms and reinforcement.

The LCS foundation (geomembrane or low-permeability soil liner) should be inspected and surveyed upon its completion to ensure that it has proper grading and is free of debris and liquids. During construction, the following activities, as appropriate, should be observed and documented:

- Pipe bedding placement including quality, thickness, and areal coverage;
- Granular filter layer placement including material quality and thickness;
- Pipe installation including location, configuration, grades, joints, filter layer placement, and final flushing;
- Granular drainage layer placement including protection of underlying liners, thickness, overlap with filter fabrics and geonets if applicable, and weather conditions;
- Geonet placement including layout, overlap, and protection from clogging by granular material carried by wind or run-off during construction;
- Geotextile/geofabric placement including coverage and overlap;
- Sumps and structure installation; and
- Mechanical and electrical equipment installations including testing.

In addition to field observations, actual field and laboratory testings may be performed to document that the materials meet the design specifications. These activities should be documented and should include the following:

- Geonet and geotextile sampling and testing;
- Granular drainage and filter layer sampling and testing for grain size distribution; and
- Testing of pipes for leaks, obstructions, and alignments.

Upon completion of construction, each component should be inspected to identify any damage that may have occurred during its installation, or during construction of another component (i.e. pipe crushing during placement of granular drainage layer). Any damage that does occur should be repaired, and these corrective measures should be documented in the CQA records.

4.4. *Relevant Point of Compliance*

The relevant point of compliance should not more than 150 m from the waste management unit boundary and must locate on lands that are owned by the MSWLF unit. In determining the relevant point of compliance, the following eights factors should be considered:

1. The hydrogeologic characteristics of the facility and surrounding land;
2. The volume and physical and chemical characteristics of the leachate;
3. The quantity, quality, and direction of flow of groundwater;
4. The proximity and withdrawal rate of the groundwater users;
5. The availability of alternative drinking-water supplies;

6. The existing quality of the groundwater, including other sources of contamination and their cumulative impacts on the groundwater and whether the groundwater is currently used or reasonably expected to be used for drinking water;
7. Public health, safety, and welfare effects; and
8. Practicable capability of the owner or operator.

However, in some limited cases, an alternative relevant point of compliance is set at the waste management unit boundary. The waste management unit boundary is defined as the vertical surface located at the hydraulically down-gradient limit of the unit. This vertical surface extends down into and through the entire thickness of the uppermost aquifer. At least above eight factors should be considered in establishing the relevant point of compliance for any design of MSWLF units. The factors provide information needed to determine if the alternative boundary is sufficiently protective of human health and the environment and if the relevant point of compliance is adequate to measure the performance of the disposal unit.

4.4.1. *Site Hydrogeology*

The first factor to be considered when determining the relevant point of compliance is site hydrogeology. Site hydrogeologic characteristics should be used to identify additional information required to set the relevant point of compliance. The site data should be sufficient to determine the lateral well-spacing required to detect contaminant releases to the uppermost aquifer.

4.4.2. *Leachate Volume and Physical Characteristics*

Data on leachate volume and quality are needed to make a determination of the "detectability" of leakage from the facility at the relevant point of compliance. The net concentration at any given point resulting from the transport of contaminants from the landfill is a function of contaminant type, initial contaminant concentration, and leakage rate that are already discussed in earlier sections.

4.4.3. *Quality, Quantity, and Direction of Groundwater Flow*

The hydrogeologic data collected should provide information to assess the groundwater flow rate, groundwater flow direction, and the volume of groundwater flow. Background groundwater quality data should be used to establish baseline concentrations of the monitoring constituents. This information will be required as input to determine if contaminants from the landfill unit have been released and have migrated to the relevant point of compliance.

4.4.4. *Groundwater Receptors*

The goal of establishing the relevant point of compliance is to ensure early detection of contamination of the uppermost aquifer. The distance to the relevant point of compliance should allow sufficient time for corrective measures to be implemented prior to the migration of contaminants to private or public water supply wells. Existing users of groundwater immediately down-gradient from the facility should be identified on a map. Users located at a down-gradient point where contaminants might be expected to migrate during the active life and post-closure care period of the facility should be identified.

4.4.5. *Alternative Drinking-water Supplies*

Consideration should be given to the availability of alternate drinking-water supplies in the event of a groundwater contamination problem. If the uppermost aquifer is the sole water supply source available, all reasonable efforts should be made to locate the relevant point of compliance as close as possible to the actual waste management unit boundary.

4.4.6. *Existing Groundwater Quality*

The existing groundwater quality, both up-gradient and down-gradient of the MSWLF unit, should be determined prior to establishing the relevant point of compliance. The performance standard for landfill design requires that landfill units be designed, so that the concentrations listed in Table 10 are not exceeded at a relevant point of compliance. However, issues need to consider whether the groundwater is currently used or is reasonably expected to be used as a drinking-water source when setting a relevant point of compliance. If the groundwater is not currently or reasonably expected to be used for drinking-water, then the relevant point of compliance to be set near the 150-m limit.

4.4.7. *Public Health, Welfare, and Safety*

Consideration should be given to the potential overall effect on public health, welfare, and safety of the proposed relevant point of compliance. Issues that should be considered include:

- Distance to the nearest groundwater user or potentially affected surface water;
- The response time (based on the distance to the proposed relevant point of compliance) required to identify and remediate or otherwise contain groundwater that may become impacted and potentially affect downgradient water supplies; and

- The risk that detection monitoring data may not be representative of a worst case release of contaminants to groundwater.

4.4.8. *Practicable Capability of the Owner or Operator*

If the relevant point of compliance is placed farther from the waste management unit boundary, the volume of water requiring treatment, should the groundwater become contaminated, will increase. One or more of the following conditions could affect the owner's or operator's practicable capability (i.e. technical and financial) to remediate contaminant releases:

- Area of impact, remedial costs, scope of remedial investigation, and site characterization;
- Increased response time due to higher costs and increased technical scope of selected remedial method;
- A reduction of the removal efficiency of treatment technologies; and
- Increased difficulty in groundwater extraction or containment if these technologies are chosen.

5. Groundwater Monitoring and Corrective Action

5.1. *General Overview*

Groundwater monitoring and corrective action is essential requirements for all existing and new MSWLF units and lateral expansions of existing units due to potential migration of leachate and or contaminant. If there is no potential migration of leachate constituents from the unit to the uppermost aquifer, then monitoring and corrective action can be suspended. These groundwater monitoring and corrective action criteria include the location selection, design, and installation of groundwater monitoring systems and set standards for groundwater sampling and analysis. In addition, it requires specific statistical methods and decision criteria for identifying a significant change in groundwater quality. If a significant change in groundwater quality occurs, the criteria require an assessment of the nature and extent of contamination followed by an evaluation and implementation of remedial measures.

5.2. *Applicability, Compliance Schedule, and Qualifications*

5.2.1. *Applicability*

Site-specific information must include, at a minimum, the information necessary to evaluate or interpret the effects of the following properties or processes on contaminant fate and transport:

5.2.1.1. Physical properties or processes

- *Aquifer Characteristics* — including hydraulic conductivity, hydraulic gradient, effective porosity, aquifer thickness, degree of saturation, stratigraphy, degree of fracturing and secondary porosity of soils and bedrock, aquifer heterogeneity, groundwater discharge, and groundwater recharge areas;
- *Waste Characteristics* — including quantity, type, and origin (i.e. commercial, industrial, or small quantity generators of unregulated hazardous wastes);
- *Climatic Conditions* — including annual precipitation, leachate generation estimates, and effects on leachate quality;
- *Leachate Characteristics* — including leachate composition, solubility, density, the presence of immiscible constituents, Eh, and pH; and
- *Engineered Controls* — including liners, cover systems, and aquifer controls (e.g. lowering the water table). These should be evaluated under design and failure conditions to estimate their long-term residual performance.

5.2.1.2. Chemical properties or processes

- *Attenuation of Contaminants in the Subsurface* — including adsorption/desorption reactions, ion exchange, organic content of soil, soil water pH, and consideration of possible reactions causing chemical transformation or chelation.

5.2.1.3. Biological processes

- *Microbiological Degradation* — which may attenuate target compounds or cause transformations of compounds, potentially forming more toxic chemical species.

When owners or operators of MSWLF units prepare a no-migration demonstration, they must use predictions that are based on maximum contaminant migration both from the unit and through the subsurface media. Assumptions about variables affecting transport should be biased toward over-estimating transport and the anticipated concentrations. Assumptions and site-specific data that are used in the fate and transport predictions should conform with transport principles and processes, including adherence to mass-balance and chemical equilibrium limitations. Within these physicochemical limitations, assumptions should be biased toward the objective of assessing the maximum potential impact on human health and the environment. The evaluation of site-specific data and assumptions may include some of the following approaches:

- Use of the upper bound of known aquifer parameters and conditions that will maximize contaminant transport (e.g., hydraulic conductivity, effective porosity, and horizontal and vertical gradients), rather than average values.

- Use of the lower range of known aquifer conditions and parameters that tend to attenuate or retard contaminant transport (e.g., dispersivities, decay coefficients, CES, organic carbon contents, and recharge conditions), rather than average values.
- Consideration of the cumulative impacts on water quality, including both existing water quality data and cumulative health risks posed by hazardous constituents likely to migrate from the MSWLF unit and other potential or known sources.

5.2.2. *Compliance Schedule*

Compliance scheduling is a self-implementing rule establishes for owner or operators of MSWLF units, which depends on the distance of the units from drinking-water sources. However, there also another alternative schedule that may use specific site-specific information to assess the relevant risks posed by different waste management units and will allow developing as standard. Existing MSWLF units and lateral expansions less followed the rules for compliance schedule that included:

- Existing MSWLF units and lateral expansions units are in less than one mile from a drinking-water intake (surface or subsurface) must be in compliance with the groundwater monitoring requirements within three years.
- Existing MSWLF units and lateral expansions units are in greater than one mile but less than two miles from a drinking-water intake (surface or subsurface) must be in compliance with the groundwater monitoring requirements within four years.
- Existing MSWLF units and lateral expansions units are in greater than two miles from a drinking-water intake (surface or subsurface) must be in compliance with the groundwater monitoring requirements within five years.
- New MSWLF units must be in compliance with the groundwater monitoring requirements before waste can be placed in the unit for further comparison.

For most facilities, these requirements will become applicable 3–5 years after the promulgation date of the rule establishes. This period should provide sufficient time for the owner or operator to conduct site investigation and characterization studies to comply with the standard requirements. For those facilities closest to drinking-water intakes, the period provides 2–3 years to assess seasonal variability in groundwater quality. A drinking-water intake includes water supplied to a user from either a surface water or groundwater source. Site-specific information (i.e. proximity to receptors, proximity and withdrawal rate of groundwater users, waste quantity, type, containment design, and age) should enable to assess potential risk to the uppermost aquifer. The resource value of the aquifer to be monitored (i.e. groundwater quality and quantity, present and future uses, and withdrawal rate of groundwater users) also may be considered. Once groundwater monitoring has been initiated, it must continue throughout the active life and closure and post-closure care periods. The post-closure period may last up to 30 years or more after the MSWLF unit has received a final cover. In addition to establishing alternative schedules for compliance with groundwater monitoring requirements, the concern local and national authorities may establish alternative schedules for certain sampling and analysis.

In some specific cases, allow flexibility in establishing alternate groundwater monitoring compliance schedules. In setting an alternative schedule, authorities will consider potential impacts to human health and the environment. In establishing alternative schedules for installing groundwater monitoring systems at existing MSWLF units, the concern authorities may consider information including the age and design of existing facilities. Using this type of information, in conjunction with knowledge of the wastes disposed, also qualitatively assess or rank facilities based on their risk to local groundwater resources.

5.2.3. *Qualifications*

A person who conducted formal academics up to post-graduate level and sufficiently enough experience on groundwater hydrology and related field as well better understanding on natural science and engineering will capable to understand the self-implementing rule and compliance schedule. Moreover, this person will have professionalism about site characteristics and be able to evaluate groundwater flow, groundwater monitoring systems, and groundwater monitoring techniques and methods. A groundwater scientist must understand and be able to apply methods to solve solute transport problems and evaluate groundwater remedial technologies. His or her education may include undergraduate or graduate studies in hydrogeology, groundwater hydrology, engineering hydrology, water resource engineering, geotechnical engineering, geology, groundwater modeling/groundwater computer modeling, and other aspects of the natural sciences.

5.3. *Groundwater Monitoring Systems*

A groundwater monitoring system consists of both background wells and wells located at the point of compliance or waste management unit boundary (i.e. down-gradient wells). The groundwater monitoring network must be capable of detecting a release from the MSWLF unit. A sufficient number of monitoring wells must be located down-gradient of the unit and be screened at intervals in the uppermost aquifer to ensure contaminant detection. Generally, up-gradient wells are used to determine background groundwater quality. The down-gradient wells must be located at the relevant point of compliance specified by expert and concern authorities' regulations, or at the waste management unit boundary that are not in compliance with regulations. If existing physical structures obstruct well placement, the down-gradient monitoring system should be placed as close to the relevant point of compliance as possible. Wells located at the relevant point of compliance must be capable of detecting contaminant releases from the MSWLF unit to the uppermost aquifer. As discussed earlier in the section pertaining to the designation of a relevant point of compliance, the point of compliance must be no greater than 150 m from the unit boundary.

The objective of a groundwater monitoring system is to intercept groundwater that has been contaminated by leachate from the MSWLF unit. Early contaminant detection is important to allow sufficient time for corrective measures to be developed and implemented before sensitive receptors are significantly affected. To accomplish this objective, the monitoring wells should be located to sample groundwater from the uppermost aquifer at the closest practicable distance from the waste management unit boundary. An alternative distance that is protective of human health and the environment may be granted by the concern local and national authorities. Since the monitoring program is intended to operate through the post-closure period, the location, design, and installation of monitoring wells should address both existing conditions and anticipated facility development, as well as expected changes in groundwater flow. The following issues need to discuss in greater deals when developing groundwater monitoring system.

5.3.1. *Uppermost Aquifer*

Monitoring wells must be placed to provide representative groundwater samples from the uppermost aquifer. The uppermost aquifer is defined as *"the geologic formation nearest to the natural ground surface that is an aquifer, as well as lower aquifers that are hydraulically interconnected with this aquifer within the facility property boundary."* These lower aquifers may be separated physically from the uppermost aquifer by less permeable strata (i.e. having a lower hydraulic conductivity) that are often termed aquitards [*an aquitard is a less permeable geologic unit or series of closely layered units (e.g. silt, clay, or shale) that in itself will not yield significant quantities of water but will transmit water through its thickness*]. Aquitards may include thicker stratigraphic sequences of clays, shales, and dense and unfractured crystalline rocks. To be considered part of the uppermost aquifer, a lower zone of saturation must be hydraulically connected to the uppermost aquifer within the facility property boundary. Generally, the degree of communication between aquifers is evaluated by groundwater pumping tests. Methods have been devised for use in analyzing aquifer test data.

5.3.2. *Determination of Background Groundwater Quality*

The goal of monitoring-well placement is to detect changes in the quality of groundwater resulting from a release from the MSWLF unit. The natural chemical composition of groundwater is controlled primarily by the mineral composition of the geologic unit comprising the aquifer. As groundwater moves from one geologic unit to another, its chemical composition may change. To reduce the probability of detecting naturally occurring differences in groundwater quality between background and down-gradient locations, only groundwater samples collected from the same geologic unit should be compared.

Groundwater quality in areas where the geology is complex can be difficult to characterize. As a result, the rule allows the owner or operator flexibility in determining where to locate wells that will be used to establish background water quality. If the facility is new, groundwater samples collected from both up-gradient and down-gradient locations prior to waste disposal can be used to establish background water quality. The sampling should be conducted to account for both seasonal and spatial variabilities in groundwater quality.

Determining background groundwater quality by sampling wells that are not hydraulically up-gradient may be necessary where hydrogeologic conditions do not allow the owner or operator to determine which wells are hydraulically up-gradient. Additionally, background groundwater quality may be determined by sampling wells that provide groundwater samples as representative or more representative than those provided by up-gradient wells. These conditions include the following:

- The facility is located above an aquifer in which groundwater flow directions change seasonally.
- The facility is located near production wells that influence the direction of groundwater flow.
- Upgradient groundwater quality is affected by a source of contamination other than the MSWLF unit.
- The proposed or existing landfill overlies a groundwater divide or local source of recharge.
- Geologic units present at down-gradient locations are absent at up-gradient locations.
- Karst terrain or fault zones modify flow.
- Nearby surface water influences groundwater flow directions.
- Waste management areas are located close to a property boundary that is upgradient of the facility.

5.3.3. *Multi-Unit Monitoring System*

A multi-unit groundwater monitoring system does not have wells at individual MSWLF unit boundaries. Instead, an imaginary line is drawn around all of the units at the facility. In Fig. 32,[31] represented a comparison of single-unit and multi-unit systems of the monitoring wells where line constitutes the relevant

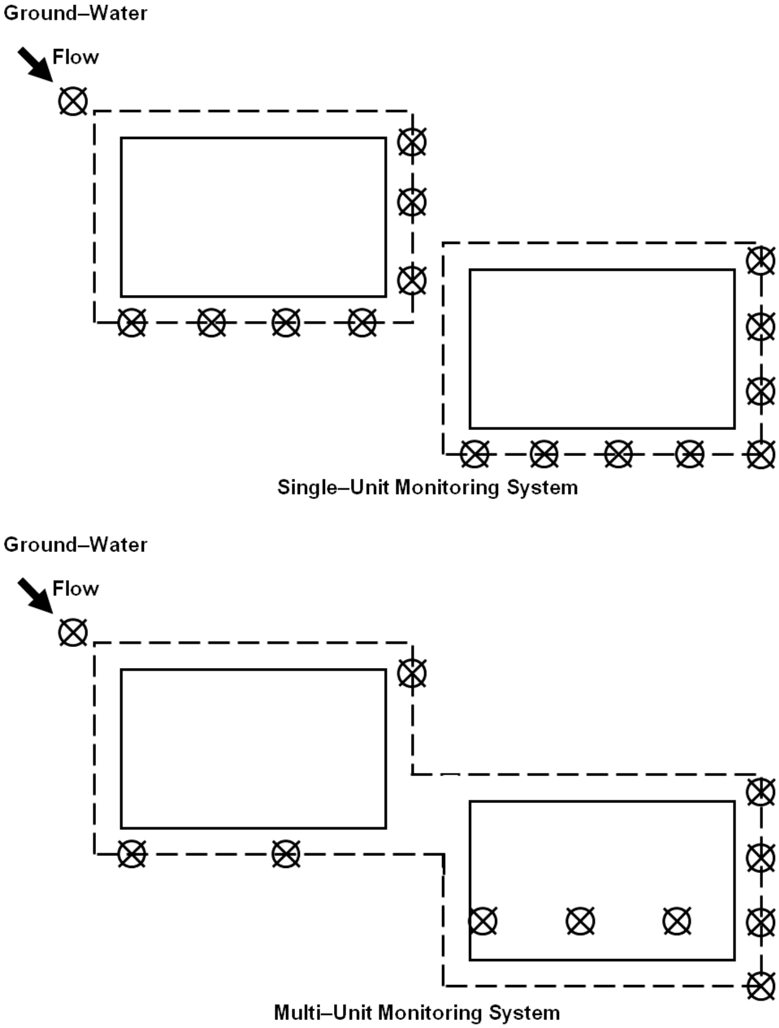


Figure 32. Comparison of Single- and Multi-unit Monitoring Systems.[37]

point of compliance. A multi-unit system must be considering the following points:

- Number, spacing, and orientation of the MSWLF units.
- Hydrogeologic setting.
- Site history.
- Engineering design of the MSWLF units.
- Type of wastes accepted at the facility.

The purpose of a multi-unit system is to reduce the number of monitoring wells that can provide the same information. The conceptual design of the multi-unit system should consider the use and management of the facility with respect to anticipated unit locations. In some cases, it may be possible to justify a reduction in the number of wells if the waste management units are aligned along the same flow path in the groundwater system. Moreover, this system must provide a level of protection to human health and the environment that is comparable to monitor individual units. Therefore, this system should allow adequate time after detection of contamination to develop and implement corrective measures before sensitive receptors are adversely affected.

5.3.4. *Hydrogeological Characteristics*

Adequate monitoring-well placement depends on collecting and evaluating hydrogeological information that can be used to form a conceptual model of the site. The goal of a hydrogeological investigation is to acquire site-specific data concerning:

- The lateral and vertical extent of the uppermost aquifer.
- The lateral and vertical extent of the upper and lower confining units/layers.
- The geology at the owner's/operator's facility (i.e. stratigraphy, lithology, and structural setting).
- The chemical properties of the uppermost aquifer and its confining layers relative to local groundwater chemistry and wastes managed at the facility.
- Groundwater flow, including:
    - — The vertical and horizontal directions of groundwater flow in the uppermost aquifer.
    - — The vertical and horizontal components of the hydraulic gradient in the uppermost and any hydraulically connected aquifer.
    - — The hydraulic conductivities of the materials that comprise the upper-most aquifer and its confining units/layers.
    - — The average linear horizontal velocity of groundwater flow in the uppermost aquifer.

Prior to initiate a field investigation, the owner or operator should perform a preliminary investigation. The preliminary investigation will involve reviewing all available information about the site, which may consist of:

- Information on the waste management history of the site, including:
    - — A chronological history of the site, including descriptions of wastes managed on-site;
    - — A summary of documented releases; and

  - Details on the structural integrity of the MSWLF unit and physical controls on waste migration.
- A literature review, including:
  - Reports of research performed in the area of the site;
  - Journal articles;
  - Studies and reports available from local, regional, and state offices (i.e. geologic surveys, water boards, and environmental agencies); and
  - Studies available from of the concern offices (like as USGS or US EPA etc.).
- Information from file searches, including:
  - Reports of previous investigations at the site and
  - Geological and environmental assessment data from local and or regional reports.

However, the documentation itemized above is by no means a complete listing of information available for a preliminary investigation. Many other sources of hydrogeological information may be available for review during the preliminary investigation.

5.3.5. *Characterizing Site Geology*

After the preliminary investigation is complete, the owner/operator will have information that can use to develop a plan to characterize site hydrogeology further. Nearly all hydrogeological investigations include a subsurface boring program. A boring program is necessary to define site hydrogeology and the small-scale geology of the area beneath the site. The program usually requires more than one iteration. The objective of the initial boreholes is to refine the conceptual model of the site derived from the preliminary investigation. The subsurface boring program should be designed as follows:

- The initial number of boreholes and their spacing is based on the information obtained during the preliminary investigation.
- Additional boreholes should be installed as needed to provide more information about the site.
- Samples should be collected from the borings at changes in lithology. For boreholes that will be completed as monitoring wells, at least one sample should be collected from the interval that will be the screened interval. Boreholes that will not be completed as monitoring wells must be properly decommissioned.

Geophysical techniques refered as cone penetrometer surveys, mapping programs, in-field probe data acquisition system, and laboratory analyses of borehole samples can be used to plan and supplement the subsurface boring program. Downhole geophysical techniques include electric, sonic, and nuclear loggings. Surface geophysical techniques include seismic reflection and refraction, as well as electromagnetic induction and resistivity. The data obtained from the subsurface boring program should enable the owner or operator to identify:

- Lithology, soil types, and stratigraphy;
- Zones of potentially high-hydraulic conductivity;
- The presence of confining formations or layers;
- Unpredicted geologic features, such as fault zones, cross-cutting structures, and pinch-out zones;
- Continuity of petrographic features, such as sorting, grain size distribution, and cementation; and
- The potentiometric surface or water table.

5.3.6. *Characterizing Groundwater Flow Beneath the Site*

In addition to characterizing site geology, the owner/operator should characterize the hydrology of the uppermost aquifer and its confining layer(s) at the site. The owner or operator should install wells and/or piezometers to assist in characterizing site hydrology. The owner/operator should determine and assess:

- The direction(s) and rate(s) of groundwater flow (including the flow of both horizontal and vertical components);
- Seasonal/temporal, natural, and artificially induced (e.g., off-site production well-pumping, agricultural use) short-term and long-term variations in groundwater elevations and flow patterns; and
- The hydraulic conductivities of the stratigraphic units at the site, including vertical hydraulic conductivity of the confining layer(s).

5.3.7. *Determining Groundwater Flow Direction and Hydraulic Gradient*

Installing monitoring wells that will provide representative background and down-gradient water samples requires a thorough understanding of how groundwater flows beneath a site. Developing such an understanding requires obtaining information regarding both groundwater flow direction(s) and hydraulic gradient. Groundwater flow direction can be thought of as the idealized path that groundwater follows as it passes through the subsurface. Hydraulic gradient (i) is the change in static head per unit of distance in a given direction. The static head is defined as the height above a standard datum of the surface of a column of water (or other liquid) that can be supported by the static pressure at a given point (i.e., the sum of the elevation head and pressure head).

To determine groundwater flow directions and hydraulic gradient, owners and operators should develop and implement a water-level-monitoring program. This program should be structured to provide precise water-level measurements in a sufficient number of piezometers or wells at a sufficient frequency to gauge both seasonal average flow directions and temporal fluctuations in groundwater flow directions. Groundwater flow direction(s) should be determined from water levels measured in wells screened in the same hydrostratigraphic position. In heterogeneous geologic settings (i.e., settings in which the hydraulic conductivities of the subsurface materials vary with location in the subsurface), long well screens can intercept stratigraphic horizons with different (i.e. contrasting) groundwater flow directions and different heads. In this situation, the resulting water levels will not provide the depth-discrete head measurements required for accurate determination of the groundwater flow direction.

In addition to evaluating the component of groundwater flow in the horizontal direction, a program should be undertaken to assess the vertical component of groundwater flow. Vertical groundwater flow information should be based, at least in part, on field data from wells and piezometers, such as multi-level wells, piezometer clusters, or multi-level sampling devices, where appropriate. The following sections provide acceptable methods for assessing the vertical and horizontal components of flow at a site.

5.3.8. *Groundwater Level Measurements*

To determine groundwater flow directions and groundwater flow rates, accurate water-level measurements (measured to the nearest 0.01 ft) should be obtained. At facilities where it is known or plausible that immiscible contaminants (i.e., non-aqueous phase liquids (NAPLs)) occur (or are determined to be potentially present after considering the waste types managed at the facility) in the subsurface at the facility, both the depth(s) to the immiscible layer(s) and the thickness(es) of the immiscible layer(s) in the well should be recorded. For the purpose of measuring total head, piezometers and wells should have as short a screened interval as possible. Specifically, the screens in piezometers or wells that are used to measure head should generally be less than 10 ft long. In circumstances including the following, well screens longer than 10 ft may be warranted:

- Natural water level fluctuations necessitate a longer screen length.
- The interval monitored is slightly greater than the appropriate screen length (e.g., the interval monitored is 12 ft thick).
- The aquifer monitored is homogeneous and extremely thick (e.g., greater than 300 ft); thus, a longer screen (i.e. a 20-ft screen) represents a fairly discrete interval.

The head measured in a well with a long screened interval is a function of all of the different heads over the entire length of the screened interval. Care should be taken when interpreting water levels collected from wells that have long screened intervals (i.e. greater than 10 ft). The water-level-monitoring program should be structured to provide precise water level measurements in a sufficient number of piezometers or wells at a sufficient frequency to gauge both seasonal average flow directions and temporal fluctuations in groundwater flow directions. The owner/operator should determine and assess seasonal/temporal, natural, and artificially induced (i.e. off-site production well-pumping, agricultural use) short- and long-term variations in groundwater elevations, groundwater flow patterns, and groundwater quality.

5.3.9. *Establishing Horizontal Flow Direction and the Horizontal Component of Hydraulic Gradient*

After the water level data and measurement procedures are reviewed to determine that they are accurate, the data should be used to:

- Construct potentiometric surface maps and water table maps based on the distribution of total head. The data used to develop water table maps should be from piezometers or wells screened across the water table. The data used to develop potentiometric surface maps should be from piezometers or wells screened at approximately the same elevation in the same hydrostratigraphic unit;
- Determine the horizontal direction(s) of groundwater flow by drawing flow lines on the potentiometric surface map or water table map (i.e., construct a flow net); and
- Calculate value(s) for the horizontal and vertical components of hydraulic gradient.

A potentiometric surface or water table map will give an approximate idea of general groundwater flow directions. However, to locate monitoring wells properly, groundwater flow direction(s) and hydraulic gradient(s) should be established in both the horizontal and vertical directions and over time at regular intervals (i.e. over a one-year period at three-month intervals).

5.3.10. *Establishing Vertical Flow Direction and the Horizontal Component of Hydraulic Gradient*

To make an adequate determination of the groundwater flow directions, the vertical component of groundwater flow should be evaluated directly. This generally requires the installation of multiple piezometers or wells in clusters or nests, or the installation of multi-level wells or sampling devices. A piezometer or well nest is a closely spaced group of piezometers or wells screened at different depths, whereas a multi-level well is a single device. Both piezometer/well nests and multi-level wells allow for the measurement of vertical variations in hydraulic head. When reviewing data obtained from multiple placements of piezometers or wells in single boreholes, the construction details of the well should be carefully evaluated. Not only it is extremely difficult to seal several piezometers/wells at discrete depths within a single borehole, but sealant materials may migrate from the seal of one piezometer/well to the screened interval of another piezometer/well. Therefore, the design of a piezometer/well nest should be considered carefully. Placement of piezometers/wells in closely spaced boreholes, where piezometers/wells have been screened at different, discrete depth intervals, is likely to produce more accurate information. The primary concerns with the installation of piezometers/wells in closely spaced, separate boreholes are:

(1) the disturbance of geologic and soil materials, which occurs when one piezometer is installed, may be reflected in the data obtained from another piezometer located nearby and
(2) the analysis of water levels measured in piezometers that are closely spaced, but separated horizontally, may produce imprecise information regarding the vertical component of groundwater flow.

The limitations of installing multiple piezometers either in single or separate boreholes may be overcome by the installation of single multi-level monitoring wells or sampling devices in single boreholes. The owner or operator should determine the vertical direction(s) of groundwater flow using the water levels measured in multi-level wells or piezometer/well nests to construct flow nets. Flow nets should depict the piezometer/well depth and length of the screened interval. It is important to portray the screened interval accurately on the flow net to ensure that the piezometer/well is actually monitoring the desired water-bearing unit. A flow net should be developed from information obtained from piezometer/well clusters or nests screened at different, discrete depths.

5.3.11. *Determining Hydraulic Conductivity*

Hydraulic conductivity is a measure of a material's ability to transmit water. A medium has a unit hydraulic conductivity if it will transmit in unit time a unit volume of groundwater at the prevailing kinematic viscosity [*Kinematic viscosity equals dynamic viscosity divided by fluid density*] through a crosssection of unit area, measured at right angles to the direction of flow, under a unit hydraulic gradient.[38] Generally, poorly sorted silty or clayey materials have low-hydraulic conductivities, whereas well-sorted sands and gravels have high-hydraulic conductivities. An aquifer may be classified as either homogeneous or heterogeneous and either isotropic or anisotropic according to the way its hydraulic conductivity varies in space. An aquifer is homogeneous if the hydraulic conductivity is independent of location within the aquifer; however, it is heterogeneous if hydraulic conductivities are dependent on location within the aquifer. If the hydraulic conductivity is independent of the direction of measurement at a point in a geologic formation, the formation is isotropic at that point. If the hydraulic conductivity varies with the direction of measurement at a point, the formation is anisotropic at that point.

5.3.11.1. Field methods

Sufficient aquifer testing (i.e., field methods) should be performed to provide representative estimates of hydraulic conductivity. Acceptable field methods include conducting aquifer tests with single wells, conducting aquifer tests with multiple wells, and using flowmeters. This section provides brief the overviews of these methods, including two methods for obtaining vertically discrete measurements of hydraulic conductivity.

A commonly used test for determining horizontal hydraulic conductivity with a single well is the slug test. A slug test is performed by suddenly adding, removing, or displacing a known volume of water from a well and observing the time that it takes for the water level to recover to its original level. Similar results can be achieved by pressurizing the well casing, depressing the water level, and suddenly releasing the pressure to simulate the removal of water from the well. In most cases, US EPA suggested that water not be introduced into wells during aquifer tests to avoid altering groundwater chemistry. Single-well tests are limited in scope to the area directly adjacent to the well screen. The vertical extent of the well screen generally defines the part of the geologic formation that is being tested. A modified version of the slug test, known as the multi-level slug test, is capable of providing depth-discrete measurements of hydraulic conductivity. The drawback of the multi-level slug test is that the test relies on the ability of the investigator to isolate a portion of the aquifer using a packer. Nevertheless, multi-level slug tests, when performed properly, can produce reliable measurements of hydraulic conductivity.

Multiple-well tests involve withdrawing water from, or injecting water into, one well and obtaining water level measurements over time in observation wells. Multiple-well tests are often performed as pumping tests in which water is pumped from one well and drawdown is observed in nearby wells. A step-drawdown test should precede most pumping tests to determine an appropriate discharge rate. Aquifer tests conducted with wells screened in the same water-bearing zone can be used to provide hydraulic conductivity data for that zone. Multiple-well tests for hydraulic conductivity characterize a greater proportion of the subsurface than single-well tests and, thus, provide average values of hydraulic conductivity. Multiple-well tests require measurement of parameters similar to those required for single-well tests (e.g., time, drawdown). When using aquifer test data to determine aquifer parameters, it is important that the solution assumptions can be applied to site conditions. Aquifer test solutions are available for a wide variety of hydrogeologic settings, but are often applied incorrectly by inexperienced persons. Incorrect assumptions regarding hydrogeology (i.e. aquifer boundaries, aquifer lithology, and aquifer thickness) may translate into incorrect estimations of hydraulic conductivity. Qualified groundwater specialist with experience in designing and interpreting aquifer tests should be consulted to ensure that aquifer test solution methods fit the hydrogeologic setting.

Multiple-well tests conducted with wells screened in different water-bearing zones furnish information concerning hydraulic communication among the zones. Water levels in these zones should be monitored during the aquifer test to determine the type of aquifer system (e.g., confined, unconfined, semi-confined, or semi-unconfined) beneath the site, and their leakance (coefficient of leakage) and drainage factors. A multiple-well aquifer test should be considered at every site as a method to establish the vertical extent of the uppermost aquifer and to evaluate hydraulic connection between aquifers. Certain aquifer tests are inappropriate for use in karst terrains characterized by a well-developed conduit flow system, and they also may be inappropriate in fractured bedrock. When a well located in a karst conduit or a large fracture is pumped, the water level in the conduit is lowered. This lowering produces a drawdown that is not radial (as in a granular aquifer) but is instead a trough-like depression parallel to the pumped conduit or fracture. Radial flow equations do not apply to drawdown data collected during such a pump test. This means that a conventional semi-log plot of drawdown versus time is inappropriate for the purpose of determining the aquifer's transmissivity and storativity. Aquifer tests in karst aquifers can be useful, but valid determinations of hydraulic conductivity, storativity, and transmissivity may be impossible. However, an aquifer test can provide information on the presence of conduits, on storage characteristics, and on the percentage of Darcian flow. Several additional factors should be considered when planning an aquifer test:

- Owners and operators should provide for the proper storage and disposal of potentially contaminated groundwater pumped from the well system.
- Owners and operators should consider the potential effects of pumping on existing plumes of contaminated groundwater.
- In designing aquifer tests and interpreting aquifer test data, owners/operators should account and correct for seasonal, temporal, and anthropogenic effects on the potentiometric surface or water table. This is usually done by installing

piezometers outside the influence of the stressed aquifer. These piezometers should be continuously monitored during the aquifer test.

- Owners and operators should be aware that, in a very high-hydraulic conductivity aquifer, the screen size and/or filter pack used in the test well can affect an aquifer test. If a very small screen size is used, and the pack is improperly graded, the test may reflect the characteristics of the filter pack, rather than the aquifer.
- In the United States, EPA recommends the use of a step-drawdown test to provide a basis for selecting discharge rates prior to conducting a full-scale pumping test. This will ensure that the pumping rate chosen for the subsequent pumping test(s) can be sustained without exceeding the available drawdown of the pumped wells. In addition, this test will produce a measurable drawdown in the observation wells.

Certain flowmeters recently have been recognized for their ability to provide accurate and vertically discrete measurements of hydraulic conductivity. One of these, the impeller flowmeter, is available commercially. More sensitive types of flowmeters (i.e., the heat-pulse flowmeter and electromagnetic flowmeter) should be available in the near future. Use of the impeller flowmeter requires running a caliper log to measure the uniformity of the diameter of the well screen. The well is then pumped with a small pump operated at a constant flow rate. The flowmeter is lowered into the well, and the discharge rate is measured every few feet by raising the flowmeter in the well. Hydraulic conductivity values can be calculated from the recorded data using the formula for horizontal flow to a well.[37–40] Use of the impeller flowmeter is limited at sites where the presence of low-permeability materials does not allow pumping of the wells at rates sufficient to operate the flowmeter. The application of flowmeters in the measure of hydraulic conductivity is described by different scientist.[37–40]

5.3.11.2. Laboratory methods

It may be beneficial to use laboratory measurements of hydraulic conductivity to augment the results of field tests. However, field methods provide the best estimates of hydraulic conductivity in most cases. Because of the limited sample size, laboratory tests can fail to account for secondary porosity features, such as fractures and joints, and hence, can greatly underestimate overall aquifer hydraulic conductivities. Laboratory tests may provide valuable information about the vertical component of hydraulic conductivity of aquifer materials. However, laboratory test results always should be confirmed by field measurements, which sample a much larger portion of the aquifer. In addition, laboratory test results can be profoundly affected by the test method selected and by the manner in which the tests are carried out (e.g., the extent to which sample collection and preparation have changed the *in situ* hydraulic properties of the tested material). Special attention should be given to the selection of the appropriate test method and test conditions and to QC of laboratory results. Laboratory tests may provide the best estimates of hydraulic conductivity for materials in the unsaturated zone, but they are likely to be less accurate than field methods for materials in the saturated zone.[37–40]

5.3.12. *Determining Groundwater Flow Rate*

The calculation of the average groundwater flow rate (average linear velocity of groundwater flow), or seepage velocity is determined by using Darcy's law and found that the average linear velocity of groundwater flow (v) is a function of hydraulic conductivity (K), hydraulic gradient (i), and effective porosity (n_{e}):

$$\bar{v} = \frac{Ki}{n_{\mathrm{e}}} \tag{7}$$

Methods for determining hydraulic gradient and hydraulic conductivity have been presented previously. Effective porosity, the percentage of the total volume of a given mass of soil, unconsolidated material, or rock that consists of interconnected pores through which water can flow, should be estimated from laboratory tests or from values cited in the literature. US EPA provides methods for determining flow rates in heterogeneous and/or anisotropic systems and should be consulted prior to calculating flow rates.[37–40]

5.3.13. *Interpreting and Presenting Hydraulic Data*

This section offers guidance on interpreting and presenting hydrogeologic data collected during the site characterization process. Graphical representations of data, such as cross sections and maps, are typically extremely helpful both when evaluating data and when presenting data to interested individuals.

5.3.13.1. Interpreting hydraulic data

Once the site characterization data have been collected, the following tasks should be undertaken to support and develop the interpretation of these data:

- Review borehole and well logs to identify major rock, unconsolidated material, and soil types and establish their horizontal and vertical extents and distributions.
- From borehole and well log (and outcrop, where available) data, construct representative cross sections for each MSWLF unit, one in the direction of groundwater flow and the other orthogonal to groundwater flow.
- Identify zones of suspected high-hydraulic conductivity, or structures likely to influence contaminant migration through the unsaturated and saturated zones.
- Compare findings with other studies and information collected during the preliminary investigation to verify the collected information.

- Determine whether or not laboratory and field data corroborate and are sufficient to define petrology, effective porosity, hydraulic conductivity, lateral and vertical stratigraphic relationships, and groundwater flow directions and rates.

After the hydrogeologic data are interpreted, the findings should be reviewed to:

- identify information gaps;
- determine whether the collection of additional data or reassessment of existing data is required to fill in the gaps; and
- identify how information gaps are likely to affect the ability to design an RCRA-monitoring system.

Generally, lithologic data should correlate with hydraulic properties (i.e. clean, well-sorted, and unconsolidated sands should exhibit high-hydraulic conductivity). Additional boreholes should be drilled and additional samples should be collected to describe the hydrogeology of the site if the investigator is unable to:

(1) correlate stratigraphic units between borings,
(2) identify zones of potentially high-hydraulic conductivity and the thickness and lateral extent of these zones, or
(3) identify confining formations/layers and the thickness and lateral extent of these formation layers.

When establishing the locations of wells that will be used to monitor groundwater in hydrogeologic settings characterized by groundwater flow in porous media, the following should be documented:

- The groundwater flow rate should be based on accurate measurements of hydraulic conductivity and hydraulic gradient and accurate measurements or estimates of effective porosity.
- The horizontal and vertical components of flow should be accurately depicted in flow nets and based on valid data.
- Any seasonal or temporal variations in the water table or potentiometric surface, and in vertical flow components, should be determined.

Once an understanding of horizontal and vertical groundwater flows has been established, it is possible to estimate where monitoring wells will most likely intercept contaminant flow.

5.3.13.2. Presenting hydraulic data

Subsequent to the generation and interpretation of site-specific geologic data, the data should be presented in geologic cross sections, topographic maps, geologic maps, and soil maps. The agency suggests that owners/operators obtain or prepare and review topographic, geologic, and soil maps of the facility, in addition to site maps of the facility and MSWLF units. In cases where suitable maps are not available, or where the information contained on available maps is not complete or accurate, detailed mapping of the site should be performed by qualified and experienced individuals. An aerial photograph and a topographic map of the site should be included as part of the presentation of hydrogeologic data. The topographic map should be constructed under the supervision of a qualified surveyor and should provide contours at a maximum of 2-ft intervals. Geologic and soil maps should be based on rock, unconsolidated material, and soil identifications gathered from borings and outcrops. The maps should use colors or symbols to represent each soil, unconsolidated material, and rock type that outcrops on the surface. The maps also should show the locations of outcrops and all borings placed during the site characterization. Geologic and soil maps are important because they can provide information describing how site geology fits into the local and regional geologic setting. Structure contour maps and isopach maps should be prepared for each water-bearing zone that comprises the uppermost aquifer and for each significant confining layer, especially the one underlying the uppermost aquifer. A structure contour map depicts the configuration (i.e. elevations) of the upper or lower surface or boundary of a particular geologic or soil formation, unit, or zone. Structure contour maps are especially important in understanding dense NAPL (DNAPL) movement because DNAPLs (i.e. tetrachloroethylene) may migrate in the direction of the dip of lower permeability units. Separate structure contour maps should be constructed for the upper and lower surfaces (or contacts) of each zone of interest. Isopach maps should depict contours that indicate the thickness of each zone. These maps are generated from borings and geologic logs and from geophysical measurements. In conjunction with cross sections, isopach maps may be used to help determine monitoring well locations, depths, and screen lengths during the design of the detection monitoring system. A potentiometric surface map or water table map should be prepared for each water-bearing zone that comprises the uppermost aquifer. Potentiometric surface and water table maps should show both the direction and rate of groundwater flow and the locations of all piezometers and wells on which they are based. The water level measurements for all piezometers and wells on which the potentiometric surface map or water table map is based should be shown on the potentiometric surface or water table map. If seasonal or temporal variations in groundwater flow occur at the site, a sufficient number of potentiometric surface or water table maps should be prepared to show these variations. Potentiometric surface and water table maps can be combined with structure contour maps for a particular formation or unit. An adequate number of cross sections should be prepared to depict significant stratigraphic and structural trends and to reflect stratigraphic and structural features in relation to local and regional groundwater flows.

5.3.13.3. Hydrological report

The hydrogeological report should contain, at a minimum:

- A description of field activities;
- Drilling and/or well construction logs;
- Analytical data;
- A discussion and interpretation of the data; and
- Recommendations to address data gaps.

The final output of the site characterization phase of the hydrogeological investigation is a conceptual model. This model is the integrated picture of the hydrogeologic system and the waste management setting. The final conceptual model must be a site-specific description of the unsaturated zone, the uppermost aquifer, and its confining units. The model should contain all of the information necessary to design a groundwater monitoring system.

5.3.14. *Monitoring Well Placement*

This section addresses the lateral placement and the vertical sampling intervals of point of compliance wells. However, these two aspects of well placement should be evaluated together in the design of the monitoring system. Site-specific hydrogeologic data obtained during the site characterization should be used to determine the lateral placement of detection monitoring wells and to select the length and vertical position of monitoring well intakes. Potential pathways for contaminant migration are 3D. Consequently, the design of a detection monitoring network that intercepts these potential pathways requires a 3D approach.

5.3.14.1. Lateral placement of point of compliance monitoring wells

Point of compliance monitoring wells should be as close as physically possible to the edge of the MSWLF unit(s) and should be screened in all transmissive zones that may act as contaminant transport pathways. The lateral placement of monitoring wells should be based on the number and spatial distribution of potential contaminant migrating pathways and on the depths and thicknesses of stratigraphic horizons that can serve as contaminant migration pathways. Point of compliance monitoring wells should be placed laterally along the downgradient edge of the MSWLF unit to intercept potential pathways for contaminant migration. The local groundwater flow direction and gradient are the major factors in determining the lateral placement of point of compliance wells. In a homogeneous and isotropic hydrogeologic setting, well placement can be based on general aquifer characteristics (e.g., direction and rate of groundwater flow), and potential contaminant fate and transport characteristics (e.g., advection and dispersion). More commonly, however, geology is variable and preferential pathways exist that control the migration of contaminants. These types of heterogeneous and anisotropic geologic settings can have numerous and discrete zones within that contaminants may migrate.

Potential migration pathways include zones of relatively high-intrinsic (matrix) hydraulic conductivities, fractured/faulted zones, and subsurface material that may increase in hydraulic conductivity if the material is exposed to waste(s) managed at the site (e.g., a limestone layer that underlies an acidic waste). In addition to natural hydrogeologic features, human-made features may influence the groundwater flow direction and, thus, the lateral placement of point of compliance wells. Such human-made features include ditches, areas where fill material has been placed, buried piping, buildings, LCSs, and adjacent disposal units. The lateral placement of monitoring wells should be based on the number and spatial distribution of potential contaminant migration pathways and on the depths and thicknesses of stratigraphic horizons that can serve as contaminant migration pathways. In some settings, the groundwater flow direction may reverse seasonally (depending on precipitation), change as a result of tidal influences or river and lake stage fluctuations, or change temporally as a result of well-pumping or changing land use patterns. In other settings, groundwater may flow away from the waste management area in all directions. In such cases, monitoring wells be installed on all sides (or in a circular pattern) around the waste management area to allow for the detection of contamination. In these cases, certain wells may be downgradient only part of the time, but such a configuration should ensure that releases from the unit will be detected.

The lateral placement of monitoring wells also should be based on the physical/chemical characteristics of the contaminants of concern. While the restriction of liquids in MSWLFs may limit the introduction of hazardous constituents into landfills, it is important to consider the physical/chemical characteristics of contaminants when designing the well system. These characteristics include solubility, Henry's law constant, partition coefficients, specific gravity, contaminant reaction or degradation products, and the potential for contaminants to degrade confining layers. For example, contaminants with low solubilities and high specific gravities that occur as DNAPLs may migrate in the subsurface in directions different from the direction of groundwater flow. Therefore, in situations where the release of DNAPLs is a concern, the lateral placement of compliance point groundwater monitoring wells should not necessarily only be along the downgradient edge of the MSWLF unit. Considering both contaminant characteristics and hydrogeologic properties is important when determining the lateral placement of monitoring wells.

5.3.14.2. Vertical placement and screen lengths

Proper selection of the vertical sampling interval is necessary to ensure that the monitoring system is capable of detecting a release from the MSWLF unit. The vertical position and lengths of well intakes are functions of:

(1) hydrogeologic factors that determine the distribution of, and fluid/vapor phase transport within, potential pathways of contaminant migration to and within the uppermost aquifer and
(2) the chemical and physical characteristics of contaminants that control their transport and distribution in the subsurface.

Well intake length also is determined by the need to obtain vertically discrete groundwater samples. Owners and operators should determine the probable location, size, and geometry of potential contaminant plumes when selecting well intake positions and lengths.

Site-specific hydrogeologic data obtained during the site characterization should be used to select the length and vertical position of monitoring well intakes. The vertical positions and lengths of monitoring well intakes should be based on the number and spatial distribution of potential contaminant migration pathways and on the depths and thicknesses of stratigraphic horizons that can serve as contaminant migration pathways. Figure 33[37] illustrated examples of complex stratigraphy that would require multiple vertical monitoring intervals.

The depth and thickness of a potential contaminant migration pathway can be determined from soil, unconsolidated material, and rock samples collected during the boring program, and from samples collected while drilling the monitoring well. Direct physical data can be supplemented by geophysical data, available regional/local hydrogeological data, and other data that provide the vertical distribution of hydraulic conductivity. The vertical sampling interval is not necessarily synonymous with aquifer thickness. Monitoring wells are often screened at intervals that represent a portion of the thickness of the aquifer. When monitoring an unconfined aquifer, the well screen typically should be positioned, so that a portion of the well screen is in the saturated zone and a portion of the well screen is in the unsaturated zone (i.e., the well screen straddles the water table). While the restriction of liquids in MSWLFs may limit the introduction of hazardous constituents into landfills, it is important to consider the physical/chemical characteristics of contaminants when designing the well system.

The vertical positions and lengths of monitoring well intakes should be based on the same physical/chemical characteristics of the contaminants of concern that influence the lateral placement of monitoring wells. Considering both contaminant characteristics and hydrogeologic properties is important when choosing the vertical position and length of the well intake. Some contaminants may migrate within very narrow zones. Of course, for well placement at a new site, it is unlikely that the owner or operator will be able to assess contaminant characteristics. Different transport processes control contaminant migration depending on whether the contaminant

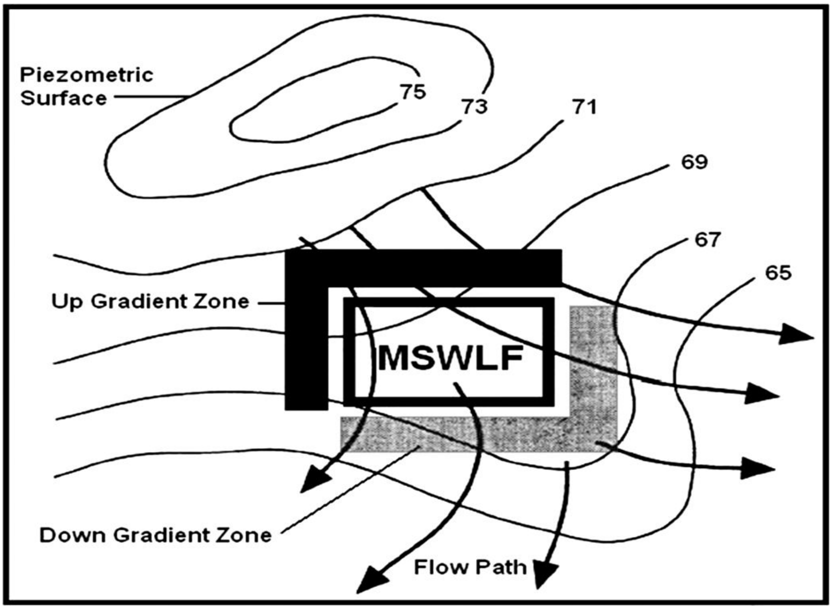


Figure 33. Upgradient and Downgradient Designations for Idealized MSWLF.[37]

dissolves or is immiscible in water. Immiscible contaminants may occur as light NAPLs (LNAPLs), which are lighter than water, and DNAPLs, which are denser than water. LNAPLs migrate in the capillary zone just above the water table. Wells installed to monitor LNAPLs should be screened at the water table/capillary zone interface, and the screened interval should intercept the water table at its minimum and maximum elevations. LNAPLs may become trapped in residual form in the vadose zone and become periodically remobilized and contribute further to aquifer contamination, either as free phase or dissolved phase contaminants, as the water table fluctuates and precipitation infiltrates the subsurface.

The migration of free-phase DNAPLs may be influenced primarily by the geology, rather than the hydrogeology, of the site. That is, DNAPLs migrate downward through the saturated zone due to density and then migrate by gravity along less permeable geologic units (i.e. the slope of confining units, the slope of clay lenses in more permeable strata, and bedrock troughs), even in aquifers with primarily horizontal groundwater flow. Consequently, if wastes disposed at the site are anticipated to exist in the subsurface as a DNAPL, the potential DNAPL should be monitored:

- At the base of the aquifer (immediately above the confining layer).
- In structural depressions (e.g., bedrock troughs) in lower hydraulic conductivity geologic units that act as confining layers.

- Along lower hydraulic conductivity lenses and units within units of higher hydraulic conductivity.
- “Down-the-dip” of lower hydraulic conductivity units that act as confining layers, both up-gradient and down-gradient of the waste management area.

Because of the nature of DNAPL migration (i.e. along structural, rather than hydraulic, gradients), wells installed to monitor DNAPLs may need to be installed both up-gradient and down-gradient of the waste management area. It may be useful to construct a structure contour map of lower permeability strata and identify lower permeability lenses upgradient and downgradient of the unit along which DNAPLs may migrate. The wells can then be located accordingly.

The lengths of well screens used in groundwater monitoring wells can significantly affect their ability to intercept releases of contaminants. The complexity of the hydrogeology of a site is an important consideration when selecting the lengths of well screens. Most hydrogeologic settings are complex (heterogeneous and anisotropic) to a certain degree. Highly heterogeneous formations require shorter well screens to allow sampling of discrete portions of the formation that can serve as contaminant migration pathways. Well screens that span more than a single saturated zone or a single contaminant migration pathway may cause cross-contamination of transmissive units, thereby increasing the extent of contamination. Well intakes should be installed in a single saturated zone. Well intakes (i.e. screens) and filter pack materials should not interconnect, or promote the interconnection of, zones that are separated by a confining layer. Even in hydrologically simple formations, or within a single potential pathway for contaminant migration, the use of shorter well screens may be necessary to detect contaminants concentrated at particular depths. A contaminant may be concentrated at a particular depth because of its physical/chemical properties and/or because of hydrogeologic properties. In homogeneous formations, a long well screen can permit excessive amounts of uncontaminated formation water to dilute the contaminated groundwater entering the well. At best, dilution can make contaminant detection difficult; at worst, contaminant detection is impossible if the concentrations of contaminants are diluted to levels below the detection limits for the prescribed analytical methods. The use of shorter well screens allows for contaminant detection by reducing excessive dilution. When placed at depths of predicted preferential flow, shorter well screens are effective in monitoring the aquifer or the portion of the aquifer of concern.

Generally, screen lengths should not exceed 10 ft. However, certain hydrogeologic settings may warrant or necessitate the use of longer well screens for adequate detection monitoring. Unconfined aquifers with widely fluctuating water tables may require longer screens to intercept the water table surface at both its maximum and minimum elevations and to provide monitoring for the presence of contaminants that are less dense than water. Saturated zones that are slightly greater in thickness than the appropriate screen length (i.e. 12 ft thick) may warrant monitoring with longer screen lengths. Extremely thick homogeneous aquifers (i.e. >300 ft) may be monitored with a longer screen (i.e. a 20-ft screen) because a slightly longer screen would represent a fairly discrete interval in a very thick formation. Formations with very low hydraulic conductivities also may require the use of longer well screens to allow sufficient amounts of formation water to enter the well for sampling. The importance of accurately identifying such conditions highlights the need for a complete hydrogeologic site investigation prior to the design and placement of detection wells.

Multiple monitoring wells (i.e. well clusters or multi-level sampling devices) should be installed at a single location when:

(1) a single well cannot adequately intercept and monitor the vertical extent of a potential pathway of contaminant migration,
(2) there is more than one potential pathway of contaminant migration in the subsurface at a single location, or
(3) there is a thick saturated zone and immiscible contaminants are present or are determined to be potentially present after considering waste types managed at the facility.

Conversely, at sites where groundwater may be contaminated by a single contaminant, where there is a thin saturated zone, and where the site is hydrogeologically homogeneous, the need for multiple wells at each sampling location is reduced. The number of wells that should be installed at each sampling location increases with site complexity.

5.4. *Groundwater Monitoring Well Design and Construction*

Groundwater monitoring well design, installation, and maintenance are basic requirements for any existing unit or lateral expansions of unit or new MSWLF units. The design, installation, and decommissioning of any monitoring well must be documented in the operating record of the facility and certified by a qualified groundwater scientist. Documentation is required for wells, piezometers, sampling devices, and water level measurement instruments used in the monitoring program. The design, installation, and maintenance of monitoring wells will affect the consistency and accuracy of samples collected. The design must be based on site-specific information. The formation material (lithology and grain size distribution) will determine the selection of proper packing and sealant materials and the stratigraphy will determine the screen length for the interval to be monitored. Installation practices should be specified and overseen to ensure that the monitoring well is installed as designed and will perform as intended. This section will discuss the factors that must be considered when designing monitoring wells. Each well must be tailored to suit the hydrogeological setting, the contaminants to be monitored, and other site-specific factors. Figure 34[37] depicts the components of a typical monitoring well installation.

5.4.1. *Selection of Drilling Method*

The method chosen for drilling a monitoring well depends largely on the following factors:

- Versatility of the drilling method;
- Relative drilling cost;
- Sample reliability (groundwater, soil, unconsolidated material, or rock samples);
- Availability of drilling equipment;
- Accessibility of the drilling site;
- Relative time required for well installation and development;
- Ability of the drilling technology to preserve natural conditions;
- Ability to install a well of desired diameter and depth; and
- Relative ease of well completion and development, including the ability to install the well in the given hydrogeologic setting.

In addition to these factors, it includes matrices to assist in selecting an appropriate drilling method. These matrices list the most commonly used drilling techniques for monitoring well installation, taking into consideration hydrogeologic settings, and the objectives of the monitoring program. The following basic performance objectives should guide the selection of drilling procedures for installing monitoring wells:

- Drilling should be performed in a manner that preserves the natural properties of the subsurface materials.
- Contamination and/or cross-contamination of groundwater and aquifer materials during drilling should be avoided.
- The drilling method should allow for the collection of representative samples of rock, unconsolidated materials, and soil.
- The drilling method should allow the owner/operator to determine when the appropriate location for the screened interval has been encountered.
- The drilling method should allow for proper placement of the filter pack and annular sealants. The borehole should be at least 4 in. larger in diameter than the nominal diameter of the well casing and screen to allow adequate space for placement of the filter pack and annular sealants.

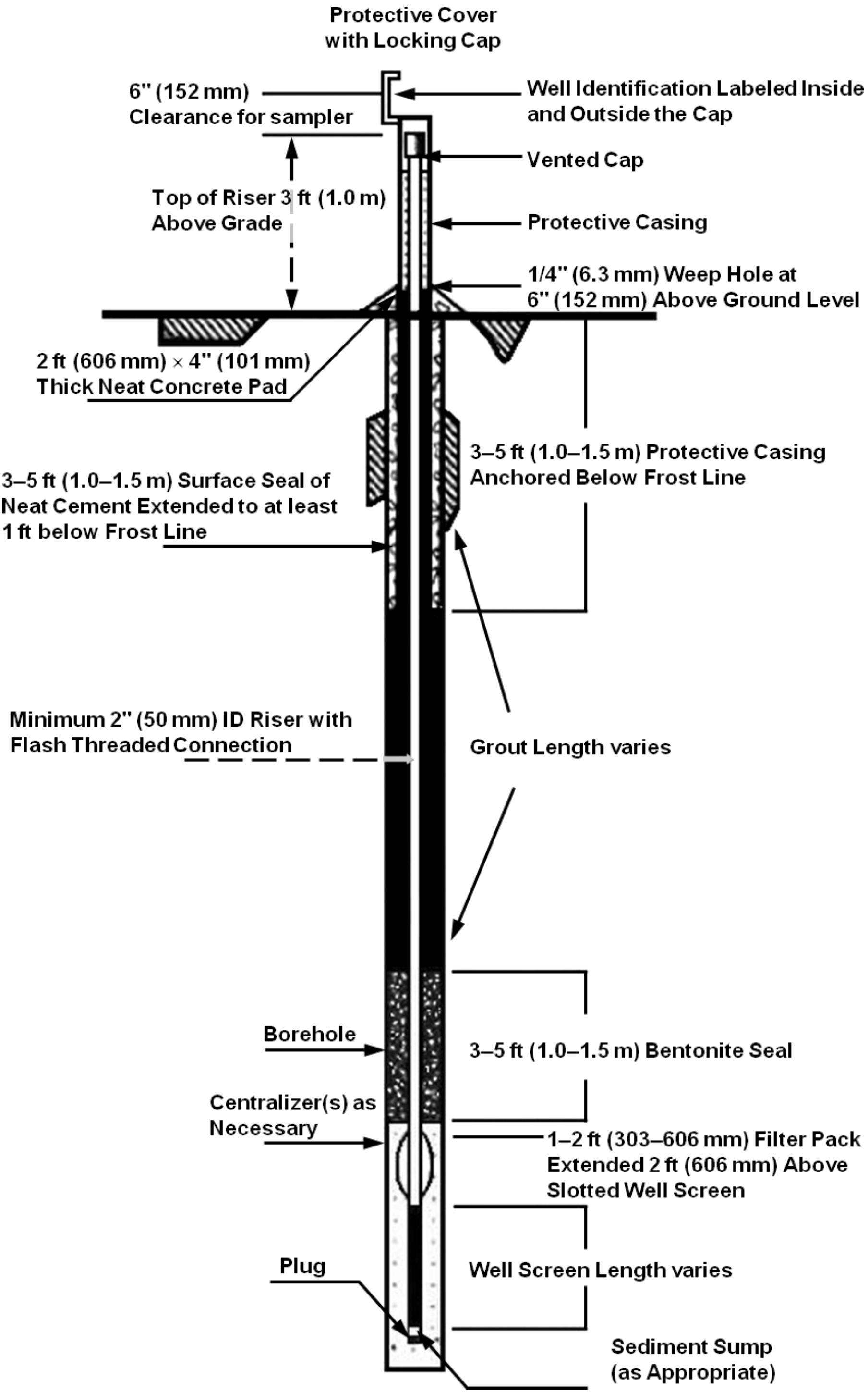


Figure 34. Monitoring Well Design-single Cased Well.[37]

- The drilling method should allow for the collection of representative groundwater samples. Drilling fluids (including air) should be used only when minimal impact to the surrounding formation and groundwater can be ensured.

The following guidelines apply to the use of drilling fluids, drilling fluid additives, and lubricants when drilling groundwater monitoring wells:

- Drilling fluids, drilling fluid additives, or lubricants that affect the analysis of hazardous constituents in groundwater samples should not be used.
- The owner/operator should demonstrate the inertness of drilling fluids, drilling fluid additives, and lubricants by performing analytical testing of drilling fluids, drilling fluid additives, and lubricants and/or by providing information regarding the composition of drilling fluids, drilling fluid additives, or lubricants obtained from the manufacturer.
- The owner/operator should consider the potential impact of drilling fluids, drilling fluid additives, and lubricants on the physical and chemical characteristics of the subsurface and on groundwater quality.
- The volume of drilling fluids, drilling fluid additives, and lubricants used during the drilling of a monitoring well should be recorded.

5.4.2. *Monitoring Well Design*

5.4.2.1. Well casing

5.4.2.1.1. *Well casing and screen materials*

A casing and well screen are installed in a groundwater monitoring well for several reasons: to provide access from the surface of the ground to some point in the subsurface, to prevent borehole collapse, and to prevent hydraulic communication between zones within the subsurface. In some cases, state or local regulations may specify the casing and material that the owner or operator should use. A comprehensive discussion of well casing and screen materials is provided in US EPA documentations. Monitoring well casing and screen materials may be constructed of any of several types of materials, but should meet the following performance specifications:

- Monitoring well casing and screen materials should maintain their structural integrity and durability in the environment in which they are used over their operating life.
- Monitoring well casings and screens should be resistant to chemical and microbiological corrosion and degradation in contaminated and uncontaminated waters.
- Monitoring well casings and screens should be able to withstand the physical forces acting upon them during and following their installation and during their

use — including forces due to suspension in the borehole, grouting, development, purging, pumping, and sampling and forces exerted on them by the surrounding geologic materials.
- Monitoring well casing and screen materials should not chemically alter groundwater samples, especially with respect to the analytes of concern, as a result of their sorbing, desorbing, or leaching analytes. For example, if chromium is an analyte of interest, the well casing or screen should not increase or decrease the amount of chromium in the groundwater. Any material leaching from the casing or screen should not be an analyte of interest or interfere in the analysis of an analyte of interest.

In addition, monitoring well casing and screen materials should be relatively easy to install into the borehole during construction of the monitoring well. The selection of the most suitable well casing and screen materials should consider site-specific factors, including:

- Depth to the water-bearing zone(s) to be monitored and the anticipated well depth.
- Geologic environment.
- Geochemistry of soil, unconsolidated material, and rock over the entire interval in which the well is to be cased.
- Geochemistry of the groundwater at the site, as determined through an initial analysis of samples.
- From both background wells and down-gradient wells and including:
    - Natural groundwater geochemistry,
    - Nature of suspected or known contaminants, and
    - Concentration of suspected or known contaminants.
- Design life of the monitoring well.

Casing materials widely available for use in groundwater monitoring wells can be divided into three categories:

(1) *Fluoropolymer materials*, including polytetrafluoroethylene (PTFE), tetrafluoroethylene (TFE), fluorinated ethylene propylene (FEP), perfluoroalkoxy (PFA), and polyvinylidene fluoride (PVDF).
(2) *Metallic materials*, including carbon steel, low-carbon steel, galvanized steel, and stainless steel (304 and 316).
(3) *Thermoplastic materials*, including PVC and acrylonitrile butadiene styrene (ABS).

In addition to these three categories of materials, fiberglass-reinforced plastic (FRP) has been used for monitoring applications. Because FRP has not yet been used in general application across the country, very little data are available on its characteristics and performance. All well construction materials possess strength-related characteristics and chemical resistance/chemical interference characteristics that influence their performance in site-specific hydrogeologic and contaminant-related monitoring situations. The casing must be made of a material strong enough to last for the life of the well. Tensile strength is needed primarily during well installation when the casing is lowered into the hole. The joint strength will determine the maximum length of a section that can be suspended from the surface in an air-filled borehole.

Collapse strength is the capability of a casing to resist collapse by any external loads to which it is subjected both during and after installation. A casing is most susceptible to collapse during installation before placement of the filter pack or annular seal materials around the casing. Once a casing is installed and supported, collapse is seldom a concern. Several steps that can be taken to avoid casing collapse are:

(1) Drilling a straight and clean borehole,
(2) Uniformly distributing filter pack materials at a slow and even rate, and
(3) Avoiding use of quick-setting (high temperature) cements for thermoplastic casings installation.

Compressive strength of the casing is a measure of the greatest compressive stress that a casing can bear without deformation. Casing failure due to a compressive strength limitation generally is not an important factor in a properly installed well. This type of failure results from soil friction on unsupported casing.

Chemical resistance/interference characteristics must be evaluated before selecting monitoring well materials. Metallic casing materials are more subject to corrosion, while thermoplastic casing materials are more susceptible to chemical degradation. The geochemistry of the formation water influences the degree to which these processes occur. If groundwater chemistry affects the structural integrity of the casing, then the samples collected from the well are likely to be affected. Materials used for monitoring well casing must not exhibit a tendency to sorb or leach chemical constituents from, or into, water sampled from the well. If a casing material sorbs constituents from groundwater, those constituents may either not be detected during monitoring or be detected at a lower concentration. Chemical constituents also can be leached from the casing materials by aggressive aqueous solutions. These constituents may be detected in samples collected from the well. The results may indicate contamination that is due to the casing rather than the formation water. Casing materials must be selected with care to avoid degradation of the well and to avoid erroneous results.

In certain situations, it may be advantageous to design a well using more than one material for well components. For example, where stainless steel or fluoropolymer materials are preferred in a specific chemical environment, costs may be saved by using PVC in noncritical portions of the well. These savings may be considerable, especially in deep wells where only the lower portion of the well is in a critical chemical environment and where tens of feet of lower-cost PVC may be used in the upper portion of the well. In a composite well design, dissimilar metallic components should not be used unless an electrically isolating design is incorporated (i.e., a dielectric coupling).

5.4.2.1.2. *Coupling procedures for joining casing*

Only a limited number of methods are available for joining lengths of casing or casing and screen together. The joining method depends on the type of casing and type of casing joint. There are generally two options available for joining metallic well casings: welding via application of heat or threaded joints. Threaded joints provide inexpensive, fast, and convenient connections and greatly reduce potential problems with chemical resistance or interference (due to corrosion) and explosion potential. Wrapping the male threads with fluoropolymer tape prior to joining sections improves the water-tightness of the joint. One disadvantage to using threaded joints is that the tensile strength of the casing string is reduced to approximately 70% of the casing strength. This reduction in strength does not usually pose a problem because strength requirements for small diameter wells (such as typical monitoring wells) are not as critical and because metallic casing has a high initial tensile strength.

Joints should create a uniform inner and outer casing diameters in monitoring well installations. Solvent cementing of thermoplastic pipe should never be used in the construction of groundwater monitoring wells. The cements used in solvent welding, which are organic chemicals, have been shown to adversely affect the integrity of groundwater samples for more than two years after well installation; only factory-threaded joints should be used on thermoplastic well material.

5.4.2.1.3. *Well casing diameter*

Although the diameter of the casing for a monitoring well depends on the purpose of the well, the casing size is generally selected to accommodate downhole equipment. Additional casing diameter selection criteria include:

(1) The drilling or well installation method used,
(2) Anticipated depth of the well and associated strength requirements,
(3) Anticipated method of well development,
(4) Volume of water required to be purged prior to sampling,
(5) Rate of recovery of the well after purging, and
(6) Anticipated aquifer testing.

5.4.2.1.4. *Casing cleaning requirements*

Well casing and screen materials should be cleaned prior to installation to remove any coatings or manufacturing residues. Prior to use, all casing and screen materials should be washed with a mild, nonphosphate, and detergent/potable water solution and rinsed with potable water. Hot pressurized water, such as in steam cleaning, should be used to remove organic solvents, oils, or lubricants from casing and screens composed of materials other than plastic. At sites where volatile organic contaminants may be monitored, the cleaning of well casing and screen materials should include a final rinse with deionized water or potable water that has not been chlorinated. Once cleaned, casings and screens should be stored in an area that is free of potential contaminants. Plastic sheeting can generally be used to cover the ground in the decontamination area to provide protection from contamination as well as should be used to clean casing and screen materials.

5.4.2.1.5. *Well intake design*

The owner/operator should design and construct the intakes of monitoring wells to:

(1) accurately sample the aquifer zone that the well is intended to sample,
(2) minimize the passage of formation materials (turbidity) into the well, and
(3) ensure sufficient structural integrity to prevent the collapse of the intake structure.

The goal of a properly completed monitoring well is to provide low-turbidity water that is representative of groundwater quality in the vicinity of the well. Close attention to proper selection of packing materials and well development procedures for wells installed in fine-grained formations (i.e. clays and silty glacial tills) is important to minimize sample turbidity from suspended and colloidal solids. There may be instances where wells completed in rock do not require screens or filter packs; the state regulatory agency should be consulted prior to completion of unscreened wells.

The selection of screen length usually depends on the objective of the well. Piezometers and wells where only a discrete flow path is monitored (such as thin gravel interbedded with clays) are generally completed using short screens (2 ft or less). To avoid dilution, the well screens should be kept to the minimum length appropriate for intercepting a contaminant plume, especially in a high-yielding aquifer. The screen length should generally not exceed 10 ft. If construction of a water table well is the objective, either for defining gradient or detecting floating phases, then a longer screen is acceptable because the owner/operator will need to provide a margin of safety that will guarantee that at least a portion of the screen always contacts the water table regardless of its seasonal fluctuations. The owner or operator should not employ well intake designs that cut across hydraulically separated geologic units.

Well screen slot size should be selected to retain from 90% to 100% of the filter pack material (discussed below) in artificially filter packed wells. Well screens should be factory-slotted. Manual slotting of screens in the field should not be performed under any circumstances.

5.4.2.2. Filter pack design

The primary filter pack material should be a chemically inert material and well rounded, with a high coefficient of uniformity. The best filter pack materials are made from industrial grade glass (quartz) sand or beads. The use of other materials, such as local and naturally occurring clean sand, is discouraged unless it can be shown that the material is inert (e.g., low CEC), coarse-grained, permeable, and uniform in grain size. The filter pack should extend at least 2 ft above the screened interval in the well.

Although design techniques for selecting filter pack size vary, all use the filter pack ratio to establish size differential between formation materials and filter pack materials. Generally, this ratio refers to either the average (50% retained) grain size of the formation material or to the 70% retained size of the formation material. However, a uniform filter pack grain size that is three to five times the size of the 50% retained size of the formation can be used. Filter pack material should be installed in a manner that prevents bridging and particle-size segregation. Filter pack material installed below the water table should generally be tremied into the annular space. Allowing filter pack material to fall by gravity (free fall) into the annular space is only appropriate when wells are relatively shallow, when the filter pack has a uniform grain size, and when the filter pack material can be poured continuously into the well without stopping.

At least 2 in. of filter pack material should be installed between the well screen and the borehole wall. The filter pack should extend at least 2 ft above the top of the well screen. In deep wells, the filter pack may not compress when initially installed. Consequently, when the annular and surface seals are placed on the filter pack, the filter pack compresses sufficiently to allow grout into, or very close to, the screen. Consequently, filter packs may need to be installed as high as 5 ft above the screened interval in monitoring wells that are deep (i.e., greater than 200 ft). The precise volume of filter pack material required should be calculated and recorded before placement, and the actual volume used should be determined and recorded during well construction. Any significant discrepancy between the calculated volume and the actual volume should be explained. Prior to installing the annular seal, a 1- to 2-ft layer of chemically inert fine sand may be placed over the filter pack to prevent the intrusion of annular or surface sealants into the filter pack. When designing monitoring wells, owners and operators should remember that the entire length of the annular space filled with filter pack material or sand is effectively the monitored zone. Moreover, if the filter pack/sand extends from the screened zone into an overlying zone, a conduit for hydraulic connection is created between the two zones.

5.4.2.3. Surface completion

Monitoring wells are commonly either above-ground completions or flush-to-ground completions. The design of both types must consider the prevention of infiltration of surface runoff into the well annulus and the possibility of accidental damage or vandalism. Completing a monitoring well involves installing the following components:

- Surface seal,
- Protective casing,
- Ventilation hole,
- Drain hole,
- Cap and lock, and
- Guard posts when wells are completed above grade.

A surface seal, installed on top of the grout seal, extends vertically up the well annulus to the land surface. To protect against frost heave, the seal should extend at least 1 ft below the frost line. The composition of the surface seal should be neat cement or concrete. In an above-ground completion, the surface seal should form at least a 2-ft wide, 4-in. thick apron. A locking protective casing should be installed around the well casing to prevent damage or unauthorized entry. The protective casing should be anchored below the frost line (where applicable) into the surface seal and extend at least 18 in. above the surface of the ground. A 1/4-in. vent hole pipe is recommended to allow the escape of any potentially explosive gases that may accumulate within the well. In addition, a drain hole should be installed in the protective casing to prevent water from accumulating and, in freezing climates, freezing around the well casing. The space between the protective casing and the well casing may be filled with gravel to allow the retrieval of tools and to prevent small animal/insect entrance through the drain. A suitable cap should be placed on the well to prevent tampering or the entry of any foreign materials. A lock should be installed on the cap to provide security. To prevent corrosion or jamming of the lock, a protective cover should be used. Care should be taken when using such lubricants as graphite or petroleum-based sprays to lubricate the lock, as lubricants may introduce a potential for sample contamination. To guard against accidental damage to the well from facility traffic, the owner/operator should install concrete or steel bumper guards around the edge of the concrete apron. These should be located within 3 or 4 ft of the well and should be painted orange or fitted with reflectors to reduce the possibility of vehicular damage. The use of flush-to-ground surface completions should be avoided because this design increases the potential for surface water infiltration into the well. In cases where flush-to-ground completions are unavoidable, such as in active roadways, a protective structure, such as a utility vault or meter box, should be installed around the well casing. In addition, measures should be taken to prevent the accumulation of surface water in the protective structure and around the well intake. These measures should include outfitting the protective structure with a steel lid or manhole cover that has a rubber seal or gasket and ensuring that the bond between the cement surface seal and the protective structure is watertight.

5.4.2.3.1. *Well surveying*

The location of all wells should be surveyed by a licensed professional surveyor (or equivalent) to determine their X- and coordinates as well as their distances from the units being monitored and their distances from each other. A state plane coordinate system, universal transverse mercator system, or latitude/longitude should be used, as approved by the regional administrator. The survey should also note the coordinates of any temporary benchmarks. A surveyed reference mark should be placed on the top of the well casing, not on the protective casing or the well apron, for use as a measuring point because the well casing is more stable than the protective casing or well apron (both the protective casing and the well apron are more susceptible to frost heave and spalling). The height of the reference survey datum, permanently marked on top of the inner well casing, should be determined within plus or minus 0.01 ft in relation to mean sea level, which in turn is determined by reference to an established national geodetic vertical datum. The reference marked on top of inner well casings should be resurveyed at least once every five years, unless changes in groundwater flow patterns/direction, or damage caused by freeze/thaw or desiccation processes, are noted. In such cases, the regional administrator may require that well casings be resurveyed on a more frequent basis.

5.4.2.3.2. *Well development*

All monitoring wells should be developed to create an effective filter pack around the well screen, to rectify damage to the formation caused by drilling, to remove fine particles from the formation near the borehole, and to assist in restoring the natural water quality of the aquifer in the vicinity of the well. Development stresses the formation around the screen, as well as the filter pack, so that mobile fines, silts, and clays are pulled into the well and removed. The process of developing a well creates a graded filter pack around the well screen. Development is also used to remove any foreign materials (drilling water, muds, etc.) that may have been introduced into the well borehole during drilling and well installation and to aid in the equilibration that will occur between the filter pack, well casing, and the formation water.

The development of a well is extremely important to ensure the collection of representative groundwater samples. If the well has been properly completed, then adequate development should remove fines that may enter the well either from the filter pack or the formation. This improves the yield, but more importantly, it creates a monitoring well capable of producing samples of acceptably low turbidity. Turbid samples from an improperly constructed and developed well may interfere with subsequent analyses. When development is initiated, a wide range of grain sizes of the natural material is drawn into the well, and the well typically produces very turbid water. However, as development continues and the natural materials are drawn into the filter pack, an effective filter will form through a sorting process. Inducing movement of groundwater into the well (i.e., in one direction) generally results in bridging of the particles. A means of inducing flow reversal is necessary to break down bridges and produce a stable filter. The commonly accepted methods for developing wells are included:

- pumping and overpumping,
- surging with a surge block, and
- bailing.

5.4.2.3.3. *Documentation of well design, construction, and development*

Information on the design, construction, and development of each well should be compiled. Such information should include:

(1) a boring log that documents well drilling and associated formation sampling and
(2) a well construction log and well construction diagram (“as built”).

5.4.2.3.4. *Decommissioning groundwater monitoring wells and boreholes*

Groundwater contamination resulting from improperly decommissioned wells and boreholes is a serious concern. Any borehole that will not be completed as a monitoring well should be properly decommissioned. Improperly constructed or unused wells should be properly decommissioned due to the following reasons:

- to eliminate physical hazards,
- to prevent groundwater contamination,
- to conserve aquifer yield and hydrostatic head, and
- to prevent intermixing of subsurface water.

Should an owner or operator have a borehole or an improperly constructed or unused well at his or her facility, the well or borehole should be decommissioned in accordance with specific guidelines. If a well to be decommissioned is contaminated, the safe removal and proper disposal of the well materials should be ensured by the owner/operator. Appropriate measures should be taken to protect the health and safety of individuals when decommissioning a well or borehole.

5.5. *Groundwater Sampling and Analysis Requirements*

The groundwater monitoring program must include consistent sampling and analysis procedures that are designed to ensure monitoring results that provide an accurate representation of groundwater quality at the background and down-gradient wells installed in compliance of monitoring requirements that conducted throughout the active life, closure, and post-closure periods. The owner's/operator's groundwater monitoring program must include a description of procedures for the following:

- sample collection,
- sample preservation and shipment,
- analytical procedures,
- chain of custody control, and
- QA and QC.

The objectives of the monitoring program should clearly define the quality of the data required to detect significant changes in groundwater chemistry due to the operation of the solid waste disposal facility. These data quality objectives should address:

- accuracy and precision of methods used in the analysis of samples, including field measurements,
- QC and QA procedures used to ensure the validity of the results (i.e. use of blank samples, record keeping, and data validation),
- number of samples required to obtain a certain degree of statistical confidence, and
- location and number of monitoring wells required.

5.5.1. *Sample Collection*

5.5.1.1. Frequency

The frequency of sample collection under detection monitoring should be evaluated for each site according to hydrogeologic conditions and landfill design. Generally, the minimum sampling frequency should be semiannual. The background characterization should include four independent samples at each monitoring location during the first semi-annual event (i.e., during the first six months of monitoring). More frequent sampling may be selected, i.e. quarterly sampling may be conducted to evaluate seasonal effects on groundwater quality. The frequency of sample collection during assessment monitoring activities will depend on site-specific hydrogeologic conditions and contaminant properties. The frequency of sampling is intended to obtain a data set that is statistically independent of the previous set.

5.5.1.2. Water level measurements

The groundwater monitoring program must include provisions for measuring static water level elevations in each well prior to purging the well for sampling. Measurements of groundwater elevations are used for determining horizontal and vertical hydraulic gradients for estimation of flow rates and direction. Field measurements may include the following:

- depth to standing water from a surveyed datum on the top of the well riser (static water level),
- total depth of well from the top of the riser (to verify condition of well), and
- thickness of immiscible layers, if present.

Measurements of the static water level and the depth to the well bottom can be made with a wetted steel tape. Electronic water level measuring devices may also be used. Accepted standard operating procedures (SOPs) call for the static water level to be accurately measured to within 0.01 ft. Water level measurements should be made at all monitoring wells and well clusters in a time frame that avoids changes that may occur because of barometric pressure changes, significant infiltration events, or aquifer pumping. To prevent possible cross contamination of wells, water-level measurement devices must be decontaminated prior to use at each well.

The groundwater monitoring program should include provisions for detecting immiscible fluids (i.e. LNAPLs or DNAPLs). LNAPLs are relatively immiscible liquids that are less dense than water and that spread across the water table surface. DNAPLs are relatively immiscible liquids that are denser than the groundwater and tend to migrate vertically downward in aquifers. The detection of an immiscible layer may require specialized equipment and should be performed before the well is evacuated for conventional sampling. The groundwater monitoring program should specify how DNAPLs and LNAPLs will be detected. The program also should include a contingency plan describing procedures for sampling and analyzing these liquids.

5.5.1.3. Well purging

Purging should be accomplished by removing groundwater from the well at low flow rates using a pump. The use of bailers to purge monitoring wells generally should be avoided. Research has shown that the "plunger" effect created by continually raising and lowering the bailer into the well can result in continual development or overdevelopment of the well. Moreover, the velocities at which groundwater enters a bailer can actually correspond to unacceptably high purging rates. The rate at which groundwater is removed from the well during purging ideally should be approximately 0.2–0.3 L/min or less. Wells should be purged at rates below those used to develop the well to prevent further development of the well, to prevent damage to the well, and to avoid disturbing accumulated corrosion or reaction products in the well. Wells also should be purged at or below their recovery rate, so that migration of water in the formation above the well screen does not occur. A low purge rate also will reduce the possibility of stripping VOCs from the water, and will reduce the likelihood of mobilizing colloids in the subsurface that are immobile under natural flow conditions. The owner/operator should ensure that purging does not cause formation water to cascade down the sides of the well screen. At no time a well should be purged to dryness if recharge causes the formation water to cascade down the sides of the screen, as this will cause an accelerated loss of volatiles. This problem should be anticipated; water should be purged from the well at a rate that does not cause recharge water to be excessively agitated. Laboratory experiments have shown that unless cascading is prevented, up to 70% of the volatiles present could be lost before sampling.

To eliminate the need to dispose of large volumes of purge water, and to reduce the amount of time required for purging, wells may be purged with the pump intake just above or just within the screened interval. This procedure eliminates the need to purge the column of stagnant water located above the well. Purging the well at the top of the well screen should ensure that freshwater from the aquifer moves through the well screen and upward within the screened interval. Pumping rates below the recharge capability of the aquifer must be maintained if purging is performed with the pump placed at the top of the well screen, below the stagnant water column above the top of the well screen. The agency suggests that a packer be placed above the screened interval to ensure that "stagnant" casing water is not drawn into the pump. The packer should be kept inflated in the well until after groundwater samples are collected.

In certain situations, purging must be performed with the pump placed at, or immediately below, the air/water interface. If a bailer must be used to sample the well, the well should be purged by placing the pump intake immediately below the air/water interface. This will ensure that all of the water in the casing and filter pack is purged, and it will minimize the possibility of mixing and/or sampling stagnant water when the bailer is lowered down into the well and subsequently retrieved. Similarly, purging should be performed at the air/water interface if sampling is not performed immediately after the well is purged without removing the pump. Pumping at the air/water interface will prevent the mixing of stagnant and freshwater when the pump used to purge the well is removed and then lowered back down into the well for the purpose of sampling.

For most wells, purging continue until the measurements of turbidity, redox potential, and dissolved oxygen in in-line or downhole analyses of groundwater have stabilized within approximately 10% over at least two measurements. If a well is purged to dryness or is purged such that full recovery exceeds two hours, the well should be sampled as soon as a sufficient volume of groundwater has entered the well to enable the collection of the necessary groundwater samples.

All purging equipment that has been or will be in contact with groundwater should be decontaminated prior to use. If the purged water or the decontamination water is contaminated (e.g., based on analytical results), the water should be stored in appropriate containers until analytical results are available, at which time proper arrangements for disposal or treatment should be made (i.e., contaminated purge water may be a hazardous waste).

5.5.1.4. Field analyses

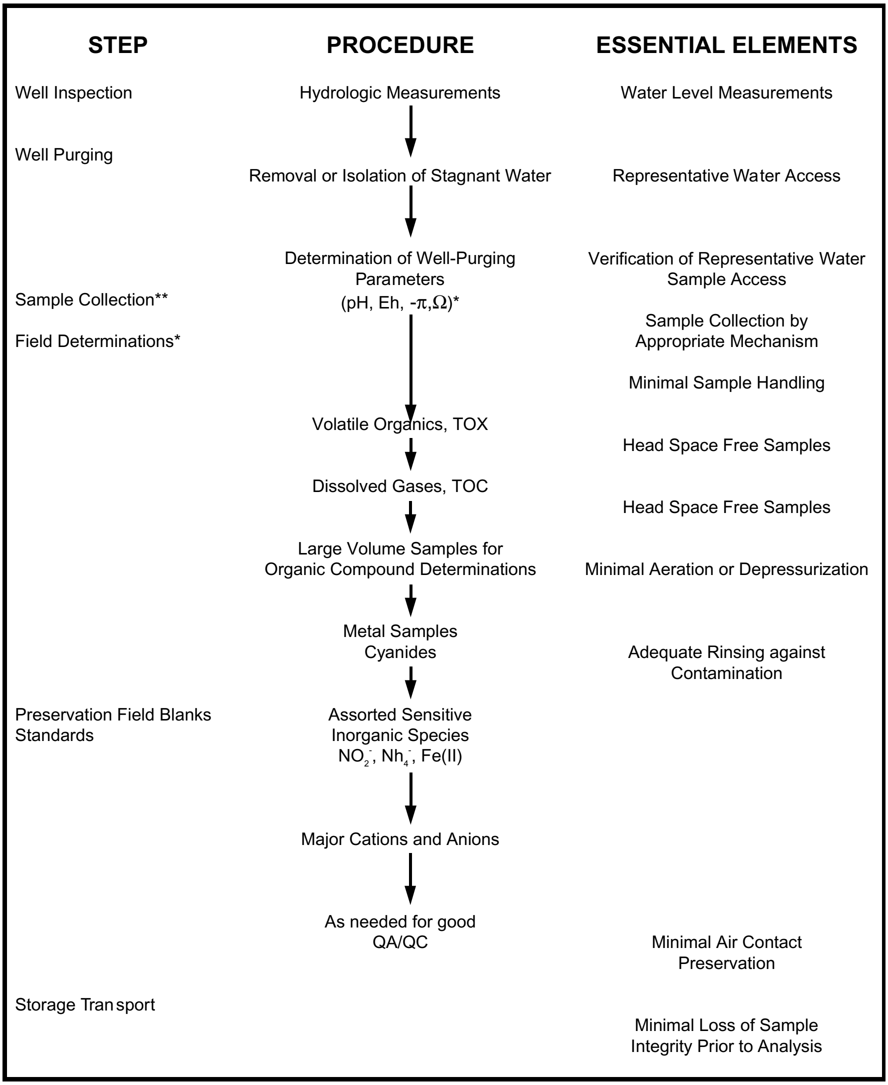
Several constituents or parameters that owners or operators may choose to include in a groundwater monitoring program may be physically or chemically unstable and should be tested after well purging and before the collection of samples for laboratory analysis. Examples of unstable parameters include pH, redox (oxidation-reduction) potential, dissolved oxygen, temperature, and specific conductance. Field analyses should not be performed on samples designated for laboratory analysis. Any field-monitoring equipment or field-test kits should be calibrated at the beginning of each use, according to the manufacturers' specifications.

5.5.1.5. Sample withdrawal and collection

The equipment used to withdraw a groundwater sample from a well must be selected based on consideration of the parameters to be analyzed in the sample. To ensure the sample is representative of groundwater in the formation, it is important to keep physical or chemical alterations of the sample to a minimum. Sampling materials and equipment should be selected to preserve sample integrity. Sampling equipment should be constructed of inert material. Sample collection equipment should not alter analyte concentrations, cause loss of analytes via sorption, or cause gain of analytes via desorption, degradation, or corrosion. Sampling equipment should be designed such that viton, tygon, silicone, or neoprene components do not come into contact with the groundwater sample. These materials have been demonstrated to cause sorptive losses of contaminants. Sorption of VOCs on silicone, polyethylene, and PVC tubing may result in gross errors when determining concentrations of trace organics in groundwater samples. The use of PVC is discourages as sampling equipment when sampling for organic contaminants. Fluorocarbon resin (i.e. Teflon) or stainless steel sampling devices that can be easily disassembled for thorough decontamination are widely used. Dedicating sampling equipment to each monitoring well will help prevent cross-contamination problems that could arise from improper decontamination procedures. Sampling equipment should cause minimal sample agitation and should be selected to reduce/eliminate sample contact with the atmosphere during sample transfer. Sampling equipment should not allow volatilization or aeration of samples to the extent that analyte concentrations are altered. Bladder pumps are generally recognized as the best overall sampling device for both organic and inorganic constituents, although other types of pumps (i.e. low-rate submersible centrifugal pumps, helical rotor electric submersible pumps, etc.) have been found suitable in some applications. Bailers, although inexpensive and simple to use, have been found to cause volatilization of samples, mobilization of particulates in wells, and imprecise results.

The following recommendations are applied to the use and operation of groundwater sampling equipment:

- Check valves should be designed and inspected to ensure that fouling problems do not reduce delivery capabilities or result in aeration of samples.
- Sampling equipment should never be dropped into the well, as this will cause degassing of the water upon impact.
- Contents of the sampling device should be transferred to sample containers in a controlled manner that will minimize sample agitation and aeration.
- Decontaminated sampling equipment should not be allowed to come into contact with the ground or other contaminated surfaces prior to insertion into the well.
- Groundwater samples should be collected as soon as possible after the well is purged. Water that has remained in the well casing for more than about 2 h has had the opportunity to exchange gases with the atmosphere and to interact with the well casing material.
- The rate at which a well is sampled should not exceed the rate at which the well was purged. Low sampling rates, approximately 0.1 L/min, are suggested. Low sampling rates will help to ensure that particulates, immobile in the subsurface under ambient conditions, are not entrained in the sample and that volatile compounds are not stripped from the sample. Pumps should be operated at rates less than 0.1 L/min when collecting samples for volatile organics analysis.
- Pump lines should be cleared at a rate of 0.1 L/min or less before collecting samples for volatiles analysis, so that the samples collected will not be from the period of time when the pump was operating more rapidly.
- Pumps should be operated in a continuous manner, so that they do not produce samples that are aerated in the return tube or upon discharge.
- When sampling wells that contain LNAPLs, a stilling tube should be inserted in the well. Groundwater samples should be collected from the screened interval of the well below the base of the tube.
- Groundwater samples collected for analysis for organic constituents or parameters should not be filtered in the field.



* Denotes analytical determinations which should be made in the field.
** This is a suggested order for sampling, not all parameters are required.

Figure 35. Generalized Flow Diagram of Groundwater Sampling Steps.[37]
Notes: *Denotes analytical determinations that should be made in the field. **This is a suggested order for sampling, not all parameters are required.

Once appropriate sampling equipment has been selected and operating procedures established, samples should be collected and containerized in the order of the volatilization sensitivity of the parameter. The preferred collection order for some of the more common groundwater analytes is depicted in the flowchart shown in Fig. 35.[37]

The groundwater monitoring program documentation should include explicit procedures for disassembly and decontamination of sampling equipment before each use. Improperly decontaminated equipment can affect samples in several ways. For example, residual contamination from the previous well may remain on equipment, or improper decontamination may not remove all of the detergents or solvents used during decontamination. Specific guidance regarding decontamination of the sampling equipment is available. To keep sample cross-contamination to a minimum, sampling should proceed from up-gradient or background locations to down-gradient locations that would contain higher concentrations of contaminants.

5.5.2. *Sample Preservation and Handling*

5.5.2.1. Sample containers

To avoid altering sample quality, the samples should be transferred from the sampling equipment directly into a prepared container. Samples should never be composited in a common container in the field and then split. Sample containers should be cleaned in a manner that is appropriate for the constituents to be analyzed. Sample containers that have been cleaned according to these procedures can be procured commercially. Most vendors will provide a certification of cleanliness.

5.5.2.2. Sample preservation

During groundwater sampling, every attempt should be made to minimize changes in the chemistry of the samples. To assist in maintaining the natural chemistry of the samples, it is necessary to preserve the sample. The owner or operator should refer to the specific preservation method and holding times for each constituent to be analyzed. Methods of sample preservation are relatively limited and are intended to retard chemical reactions, such as oxidation and biodegradation, and to reduce the effects of sorption. Preservation methods are generally limited to pH control, refrigeration, and protection from light.

5.5.2.3. Sample storage and shipment

The storage and transport of groundwater samples must be performed in a manner that maintains sample quality. Samples should be cooled to 4°C as soon as possible after they are collected. These conditions should be maintained until the samples are received at the laboratory. Sample containers generally are packed in picnic coolers or special containers for shipment. Polystyrene foam, vermiculite, and “bubble pack” are frequently used to pack sample containers to prevent breakage. Ice is placed in sealed plastic bags and added to the cooler. All related paperwork is sealed in a plastic bag and taped to the inside top of the cooler. The cooler top is then taped shut. Custody seals should be placed across the hinges and latches on the outside of the cooler. Transportation arrangements should maintain proper storage conditions and provide for effective sample pickup and delivery to the laboratory. Sampling plans should be coordinated with the laboratory, so that appropriate sample receipt, storage, analysis, and custody arrangements can be provided. Most analyses must be performed within a specified period (holding time) from sample collection. Holding time refers to the period that begins when the sample is collected from the well and ends with its extraction or analysis. Data from samples not analyzed within the recommended holding times should be considered suspect. Some holding times for hazardous constituents are as short as seven days. To provide the laboratory with operational flexibility in meeting these holding times, samples usually are shipped via overnight courier. Laboratory capacity or operating hours may influence sampling schedules. Coordination with laboratory staff during planning and sampling activities is important in maintaining sample and analysis quality. The documentation that accompanies samples during shipment to the laboratory usually includes chain-of-custody (including a listing of all sample containers), requested analyses, and full identification of the origin of samples (including contact names, phone numbers, and addresses). Copies of all documents shipped with the samples should be retained by the sampler.

5.5.3. *Chain-of-Custody Record*

To document sample possession from the time of collection, a chain-of-custody record should be filled out to accompany every sample shipment. The record should contain the following types of information:

- Sample number,
- Signature of collector,
- Date and time of collection,
- Media sampled (e.g., groundwater),
- Sample type (e.g., grab),
- Identification of sampling location/well,
- Number of containers,
- Parameters requested for analysis,
- Signatures of persons involved in the chain of possession,
- Inclusive dates of possession with time in 24-h notation,
- Internal temperature of shipping container when samples were sealed into the container for shipping,
- Internal temperature of container when opened at the laboratory, and
- Any remarks regarding potential hazards or other information the laboratory may need.

An adequate chain-of-custody program allows for tracing the possession and handling of individual samples from the time of collection through completion of laboratory analysis. A chain-of-custody program should include:

- Sample labels to prevent misidentification of samples,
- Sample custody seals to preserve the integrity of the samples from the time they are collected until they are opened in the laboratory,
- Field notes to record information about each sample collected during the groundwater monitoring program,
- Chain-of-custody record to document sample possession from the time of collection to analysis, and
- Laboratory storage and analysis records, which are maintained at the laboratory and which record pertinent information about the sample.

5.5.3.1. Sample labels

Each sample's identification should be marked clearly in waterproof ink on the sample container. To aid in labeling, the information should be written on each container prior to filling with a sample. The labels should be sufficiently durable to remain legible even when wet and should contain the following information:

- Sample identification number,
- Name and signature of the sampler,
- Date and time of collection,
- Sample location, and
- Analyses requested.

5.5.3.2. Sample custody seal

Sample custody seals should be placed on the shipping container and/or individual sample bottle in a manner that will break the seal if the container or sample is tampered with.

5.5.3.3. Field logbook

To provide an account of all activities involved in sample collection, all sampling activities, measurements, and observations should be noted in a field log. The information should include visual appearance (e.g., color, turbidity, degassing, and surface film), odor (type and strength), and field measurements and calibration results. Ambient conditions (temperature, humidity, wind, and precipitation) and well purging and sampling activities should also be recorded as an aid in evaluating sample analysis results. The field logbook should document the following:

- Well identification,
- Well depth,
- Static water level depth and measurement techniques,
- Presence and thickness of immiscible layers and the detection method,
- Well yield (high or low) and well recovery after purging (slow or fast),
- Well purging procedure and equipment,
- Purge volume and pumping rate,
- Time well purged,
- Collection method for immiscible layers,
- Sample withdrawal procedure and equipment,
- Date and time of sample collection,
- Results of field analysis,
- Well sampling sequence,
- Types of sample bottles used and sample identification numbers,
- Preservatives used,
- Parameters requested for analysis,
- Field observations of sampling event,
- Name of collector,
- Weather conditions, including air temperature, and
- Internal temperature of field and shipping containers.

5.5.3.4. Sample analysis request sheet

A sample analysis request sheet should accompany the sample(s) to the laboratory and clearly identify which sample containers have been designated for each requested parameter and the preservation methods used. The record should include the following types of information:

- Name of person receiving the sample,
- Laboratory sample number (if different from field number),
- Date of sample receipt,
- Analyses to be performed (including desired analytical method), and
- Information that may be useful to the laboratory (e.g., type and quantity of preservatives added and unusual conditions).

5.5.3.5. Laboratory records

Once the sample has been received in the laboratory, the sample custodian and/or laboratory personnel should clearly document the processing steps that are applied to the sample. All sample preparation (e.g., extraction) and determinative steps should be identified in the laboratory records. Deviations from established methods or SOPs, such as the use of specific reagents (e.g., solvents, acids), temperatures, reaction times, and instrument settings, should be noted. The results of the analyses of all QC samples should be identified for each batch of groundwater samples analyzed. The laboratory logbook should include the time, date, and name of the person who performed each processing step.

5.6. *Corrective Actions*

5.6.1. *Assessment*

An assessment of corrective measures must be conducted whenever any hazardous constituents (Table 10) are detected as exceeding MCLs of the GWPS. The assessment of corrective measures must be initiated within 90 days of the finding. During the initiation of an assessment of corrective measures, the assessment of monitoring must be also continued. The assessment of corrective measures must consider performance (including potential impacts), time, and cost aspects of the remedies. Finally, the results of the corrective measures assessment must be discussed in a public meeting with interested and affected parties. An assessment of corrective measures is site-specific and will vary significantly depending on the design and age of the facility, the completeness of the facility's historical records, the nature and concentration of the contaminants found in the groundwater, the complexity of the site hydrogeology, and the facility's proximity to sensitive receptors. Corrective measures are generally approached from two directions:

1. identify and remediate the source of contamination and
2. identify and remediate the known contamination.

Because each case will be site-specific, the owner or operator should be prepared to document that, to the best of his or her technical and financial abilities, a diligent effort has been made to complete the assessment in the shortest time practicable. Some general factors are discussed below in terms of source evaluation, plume delineation, groundwater assessment, and corrective measures assessment.

5.6.1.1. Source evaluation

As part of the assessment of corrective measures, the owner or operator will need to identify the nature of the source of the release. The first step in this identification is a review of all available site information regarding facility design, wastes received, and onsite management practices. For newer facilities, this may be a relatively simple task. However, at some older facilities, detailed records of the facility's history may not be as well documented, making source definition more difficult. Design, climatological, and waste-type information should be used to evaluate the duration of the release, potential seasonal effects due to precipitation (increased infiltration and leachate generation), and possible constituent concentrations. If source evaluation is able to identify a repairable engineering condition that likely contributed to the cause of contamination (e.g., unlined leachate storage ponds, failed cover system, leaky leachate transport pipes, past conditions of contaminated storm overflow), such information should be considered as part of the assessment of corrective measures.

Existing site geology and hydrogeology information, groundwater monitoring results, and topographic and cultural information must be documented clearly and accurately. This information may include soil boring logs, test pit and monitoring well logs, geophysical data, water level elevation data, and other information collected during facility design or operation. The information should be expressed in a manner that will aid interpretation of data. Such data may include isopach maps of the thickness of the upper aquifer and important strata, isoconcentration maps of contaminants, flow nets, cross sections, and contour maps.

5.6.1.2. Plume delineation

To effectively assess corrective measures, the lateral and vertical extents of contamination must be known. When it is determined that a GWPS is exceeded during the assessment monitoring program, it may be necessary to install additional wells to characterize the contaminant plume(s). At least one additional well must be added at the property boundary in the direction of contaminant migration to allow timely notification to potentially affected parties if contamination migrates offsite. The following circumstances may require additional monitoring wells:

- Facilities that have not determined the horizontal and vertical extents of the contaminant plume.
- Locations where the subsurface is heterogeneous or where groundwater flow patterns are difficult to establish.
- Mounding associated with MSWLF units.

Because the requirements for additional monitoring are site-specific, the regulation does not specifically establish cases where additional wells are necessary or establish the number of additional wells that must be installed. During the plume delineation process, the owner or operator is not relieved from continuing the assessment-monitoring program.

The rate of plume migration and the change in contaminant concentrations with time must be monitored to allow prediction of the extent and timing of impact to sensitive receptors. The receptors may include users of both groundwater and surface water bodies where contaminated groundwater may be discharged. In some cases, transfer of volatile compounds from groundwater to the soil and to the air may provide an additional migration pathway. Information regarding the aquifer characteristics (i.e. hydraulic conductivity, storage coefficients, and effective porosity) should be developed for modeling contaminant transport if sufficient data are not available. Anisotropy and heterogeneity of the aquifer must be evaluated, as well as magnitude and duration of source inputs, to help explain present and predicted plume configuration. Currently, most treatment options for groundwater contamination at MSWLF units involve pump and treat or *in situ* biological technologies (bioremediation). The cost and duration of treatment depend on the size of the plume, the pumping characteristics of the aquifer, and the chemical transport phenomena. Source control and groundwater flow control measures to reduce the rate of contaminant migration should be included in the costs of any remedial activity undertaken. Groundwater modeling of the plume may be initiated to establish the following:

- The locations and pumping rates of withdrawal and/or injection wells,
- Predictions of contaminant concentrations at exposure points,
- Locations of additional monitoring wells,
- The effect that source control options may have on groundwater remediation, and
- The effects of advection and dispersion, retardation, adsorption, and other attenuation processes on the plume dimensions and contaminant concentrations.

Any modeling effort must consider that simulations of remedial response measures and contaminant transport are based on many necessary simplifying assumptions, which affect the accuracy of the model. These assumptions include boundary conditions, the degree and spatial variabilities of anisotropy, dispersivity, effective porosity, stratigraphy, and the algorithms used to solve contaminant transport equations. Model selection should be appropriate for the amount of data available, and the technical uncertainty of the model results must be documented by a sensitivity analysis on the input parameters. A sensitivity analysis is generally done after model calibration by varying one input parameter at a time over a realistic range and then evaluating changes in model output.

5.6.1.3. Groundwater assessment

To assess the potential effectiveness of corrective measures for groundwater contamination, the following information is needed:

- Plume definition (includes the types, concentration, and spatial distribution of the contaminants),
- The amenability of the contaminants to specific treatment and potential for contaminants to interfere with treatability,
- Fate of the contaminants (whether chemical transformations have, are, or may be occurring, and the degree to which the species are sorbed to the geologic matrix),

- Stratigraphy and hydraulic properties of the aquifer, and
- Treatment concentration goals and objectives.

The owner or operator should consider whether immediate measures to limit further plume migration (i.e. containment options) or measures to minimize further introduction of contaminants to groundwater are necessary.

The process by which a remedial action is undertaken will generally include the following activities:

- Hydrogeologic investigation, which may include additional well installations, detailed vertical and lateral samplings to characterize the plume, and core sampling to determine the degree of sorption of constituents on the geologic matrix.
- Risk assessment, to determine the impact on sensitive receptors, which may include identification of the need to develop treatment goals other than GWPSs.
- Literature and technical reviews of treatment technologies considered for further study or implementation.
- Evaluation of costs of different treatment options.
- Estimation of the time required for completion of remediation under the different treatment options.
- Bench-scale treatability studies conducted to assess potential effectiveness of options.
- Selection of technology(ies) and proposal preparation for regulatory and public reviews and comments.
- Full-scale pilot study for verification of treatability and optimization of the selected technology.
- Initiation of full-scale treatment technology with adjustments, as necessary.
- Continuation of remedial action until treatment goals are achieved.

5.6.1.4. Corrective measures assessment

To compare different treatment options, substantial amounts of technical information must be assembled and assessed. The objective of this information-gathering task is to identify the following items for each treatment technology:

- The expected performance of individual approaches.
- The time frame when individual approaches can realistically be implemented.
- The technical feasibility of the remediation, including new and innovative technologies, performance, reliability and ease of implementation, safety, and cross media impacts.
- The anticipated time frame when remediation should be complete.

- The anticipated cost of the remediation, including capital expenditures, design, ongoing engineering, and monitoring of results.
- Technical and financial capabilities of the owner or operator to successfully complete the remediation.
- Disposal requirements for treatment residuals.
- Other regulatory or institutional requirements, including state and local permits, prohibitions, or environmental restrictions that may affect the implementation of the proposed remedial activity.

The performance objectives of the corrective measures should be considered in terms of source reduction, cleanup goals, and cleanup time frame. Source reduction would include measures to reduce or stop further releases and may include the repair of existing facility components (liner systems, leachate storage pond liners, piping systems, and cover systems), upgrading of components (liners and cover systems), or premature closure in extreme cases. The technology proposed as a cleanup measure should be the best available technology, given the practicable capability of the owner or operator. The technologies identified should be reliable, based on their previous performance; however, new innovative technologies are not discouraged if they can be shown, with a reasonable degree of confidence, to be reliable. Because most treatment processes, including biorestoration, potentially produce byproducts or release contaminants to different media (e.g., air stripping of volatile compounds), the impacts of such potential releases must be evaluated. Releases to air may constitute a worker health and safety concern and must be addressed as part of the alternative assessment process. Other cross media impacts, including transfer of contaminants from soils to groundwater, surface water, or air should be assessed and addressed in the assessment of corrective actions. Analyses should be conducted on treatment options to determine whether or not they are protective of human health and the environment. Environmental monitoring of exposure routes (air and water) may necessitate health monitoring for personnel involved in treatment activities if unacceptable levels of exposure are possible. On a case-by-case basis, implementation plans may require both forms of monitoring.

The development and screening of individual corrective measures requires an understanding of the physicochemical relationships and interferences between the constituents and the sequence of treatment measures that must be implemented. Proper sequencing of treatment methods to produce a feasible remedial program must be evaluated to avoid interference between the presence of some constituents and the effective removal of the targeted compound. In addition, screening and design parameters of potential treatment options should be evaluated in the early stages of conceptual development and planning to eliminate technically unsuitable treatment methods. In general, the selection of an appropriate treatment method will require the experience of a qualified professional and will necessitate a literature review of the best available treatment technologies. Numerous case studies and published papers from scientific and engineering technical journals exist on treatability of specific compounds and groups of related compounds. Development of new technologies and refinements of technologies have been rapid. A compendium of available literature that includes treatment technologies for organic and inorganic contaminants, technology selection, and other sources of information (i.e. literature search data bases pertinent to groundwater extraction, treatment, and responses) is included in practical guide for assessing and remediating contaminated sites (PGARCS).

The general approach to remediation typically includes active restoration, plume containment, and source control as discussed below. The selection of a particular approach or combination of approaches must be based on the corrective action objectives. It should be emphasized that the objective of a treatment program should be to restore groundwater to preexisting conditions or to levels below applicable GWPSs while simultaneously restricting further releases of contaminants to groundwater. Once treatment objectives are met, the chance of further contamination should be mitigated to the extent practicable.

5.6.1.5. Active restoration

Active restoration generally includes groundwater extraction, followed by onsite or offsite wastewater treatment. Offsite wastewater treatment may include sending the contaminated water to a local publicly owned treatment works (POTW) or to a facility designed to treat the contaminants of concern. Treated groundwater may be re-injected, sent to a local POTW, or discharged to a local body of surface water, depending on site-specific requirements. Typical treatment practices that may be implemented include coagulation and precipitation of metals, chemical oxidation of a number of organic compounds, air stripping to remove VOCs, and biological degradation of other organics.

The rate of contaminant removal from groundwater will depend on the rate of groundwater removal, the CEC of the soil, and partition coefficients of the constituents sorbed to the soil. As the concentration of contaminants in the groundwater is reduced, the rate at which constituents become partitioned from the soil to the aqueous phase may also be reduced. The amount of flushing of the aquifer material required to remove the contaminants to an acceptable level will generally determine the time frame required for restoration. This time frame is site-specific and may last indefinitely.

In situ methods may be appropriate for some sites, particularly where pump and treat technologies create serious adverse effects or where it may be financially prohibitive. *In situ* methods may include biological restoration requiring pH control, addition of specific microorganisms, and/or addition of nutrients and substrate to augment and encourage degradation by indigenous microbial populations. Bioremediation requires laboratory treatability studies and pilot field studies to determine the feasibility and the reliability of full-scale treatment. It must be demonstrated that the treatment techniques will not cause degradation of a target chemical to another compound that has unacceptable health risks and that is subject to further degradation.

Alternative *in situ* methods may also be designed to increase the effectiveness of desorption or removal of contaminants from the aquifer matrix. Such methodologies may include steam stripping, soil flushing, vapor extraction, thermal desorption, and solvent washing, and extraction for removal of strongly sorbed organic compounds. These methods also may be used in unsaturated zones where residual contaminants may be sorbed to the geologic matrix during periodic fluctuations of the water table.

5.6.1.6. Plume containment

The purpose of plume containment is to limit the spread of the contaminants. Methods to contain plume movement include passive hydraulic barriers, such as grout curtains and slurry walls, and active gradient control systems involving pumping wells and French drains. The types of aquifer characteristics that favor plume containment include:

- Water naturally unsuited for human consumption.
- Contaminants present in low concentration with low mobility.
- Low potential for exposure to contaminants and low risk associated with exposure.
- Low transmissivity and low future user demand.

Often, it may be advantageous for the owner or operator to consider implementing groundwater controls to inhibit further contamination or the spread of contamination. If groundwater pumping is considered for capturing the leading edge of the contaminant plume, the contaminated water must be managed in conformance with all applicable requirements. Under most conditions, it is necessary to consult the regulatory agencies prior to initiate an interim remedial action.

5.6.1.7. Source control

Source control measures should be evaluated to limit the migration of the plume. The regulation does not limit the definition of source control to exclude any specific type of remediation. Remedies must control the source to reduce or eliminate further releases by identifying and locating the cause of the release (i.e. torn geomembrane, excessive head due to blocked LCS, leaking leachate collection well or pipe). Source control measures may include the following:

- Modifying the operational procedures (e.g., banning specific wastes or lowering the head over the LCS through more frequent leachate removal).
- Undertaking more extensive and effective maintenance activities (e.g., excavate waste to repair a liner failure or a clogged LCS).
- Preventing additional leachate generation that may reach a liner failure (e.g., using a portable or temporary rain shelter during operations or capping landfill areas that contribute to leachate migrating from identified failure areas).

In extreme cases, the excavation of deposited wastes for treatment and/or offsite disposal may be considered.

5.6.1.8. Public participation

The owner or operator is required to hold a public meeting to discuss the results of the corrective action assessment and to identify proposed remedies. Notifications, such as contacting local public agencies, local governments, and national governments, posting a notice in prominent local newspapers, and making radio announcements are effective. The public meeting should provide a detailed discussion of how the owner or operator has addressed the factors of corrective measures.

5.6.2. *Effectiveness*

The owner or operator must consider specific topics to satisfy the performance criteria under selection of the final corrective measure. These topics must be addressed in the report documenting the selection of a particular corrective action. The general topic areas that must be considered include the following:

- The anticipated long- and short-term effectivenesses of the corrective action.
- The anticipated effectiveness of source reduction efforts.
- The ease or difficulty of implementing the corrective measure.
- The technical and economic practicable capabilities of the owner or operator.
- The degree to which the selected remedy will address concerns raised by the community.

5.6.2.1. Effectiveness of corrective action

In selecting the remedial action, the anticipated long- and short-term effectivenesses should be evaluated. Long-term effectiveness focuses on the risks remaining after corrective measures have been taken. Short-term effectiveness addresses the risks during the construction and implementation of the corrective measure. The review of case studies where similar technologies have been applied provide the best measures to judge technical uncertainty, especially when relatively new technologies are applied. The long-term and post-cleanup effectiveness may be judged on the ability of the proposed remedy to mitigate further releases of contaminants to the environment, as well as on the feasibility of the proposed remedy to meet or exceed the GWPSs. The owner or operator must make a reasonable effort to estimate and quantify risks, based on exposure pathways and estimates of exposure levels and durations. These estimates include risks for both groundwater and cross-media contaminations.

The source control measures that will be implemented, including excavation, transportation, re-disposal, and containment, should be evaluated with respect to potential exposure and risk to human health and the environment. The source control measures should be viewed as an integral component of the overall corrective action. Health considerations must address monitoring risks to workers and the general public and provide contingency plans should an unanticipated exposure occur. Potential exposure should consider both long- and short-term cases before, during, and after the implementation of corrective actions. The time to complete the remedial activity must be estimated, because it will have direct financial impacts on the project management needs and the financial capability of the owner or operator to meet the remedial objectives. The long-term costs of the remedial alternatives and the long-term financial condition of the owner or operator should be reviewed carefully. The implementation schedule should indicate QC measures to assess the progress of the corrective measure. The operational reliability of the corrective measures should be considered. In addition, the institutional controls and management practices developed to assess the reliability should be identified.

5.6.2.2. Effectiveness of source reduction

Source control measures identified in previous sections should be discussed in terms of their expected effectiveness. If source control consists of the removal and re-disposal of wastes and the residual materials, such as contaminated soils above the water table, should be quantified and their potential to cause further contamination evaluated. Engineering controls intended to upgrade or repair deficient conditions in landfill component systems, including cover systems, should be quantified in terms of anticipated effectiveness according to current and future conditions. This assessment may indicate to what extent it is technically and financially practicable to make use of existing technologies. The decision against using a certain technology may be based on health considerations and the potential for unacceptable exposure(s) to both workers and the public.

5.6.2.3. Implementation of remedial action

The ease of implementing the proposed remedial action will affect the schedule and startup success of the remedial action. The following key factors need to be assessed:

- The availability of technical expertise.
- The construction of equipment or technology.
- The ability to properly manage and dispose the wastes generated by treatment.
- The likelihood of obtaining local permits and public support for the proposed project.

Technical considerations, including pH control, groundwater extraction feasibility, or the ability to inject nutrients, may need to be considered, depending on the proposed treatment method. Potential impacts, such as potential cross-media contamination, need to be reviewed as part of the overall feasibility of the project.

The schedule of remedial activities should identify the start and end points of the following periods:

- Permitting phase.
- Construction and startup period, during which initial implementation success will be evaluated, including time to correct any unexpected problems.
- Time when full-scale treatment will be initiated and duration of treatment period.
- Implementation and completion of source control measures, including the timeframe for solving problems associated with interim management and disposal of waste materials or treatment residuals.

Items that require long lead times should be identified early in the process and those tasks should be initiated early to ensure that implementation occurs in the shortest practicable period.

5.6.2.4. Practical capability

The owner or operator must be technically and financially capable of implementing the chosen remedial alternative and ensuring project completion, including provisions for future changes to the remedial plan after progress is reviewed. If either technical or financial capability is inadequate for a particular alternative, then other alternatives with similar levels of protectiveness should be considered for implementation.

5.6.2.5. Community concerns

The public meetings held during the assessment of alternative measures are intended to elicit public comment and response. The owner or operator must, by means of meeting minutes and a record of written comments, identify which public concerns have been expressed and addressed by corrective measure options. In reality, the final remedy selected and implemented will be one that the local/national regulatory agency, the public, and the owner or operators agree to.

5.6.3. *Scheduling*

The owner or operator must consider the following factors in determining the schedule of remedial activities:

- Extent and nature of contamination.
- Practical capabilities of remedial technologies in achieving compliance with GWPSs.
- Availability of treatment or disposal capacity for wastes managed during implementation of the remedy.
- Desirability of utilizing technologies that are not currently available, but that may offer significant advantages over already available technologies in terms of effectiveness, reliability, safety, or ability to achieve remedial objectives.
- Potential risks to human health and the environment from exposure to contamination prior to completion of the remedy.
- Resource value of the aquifer including: (i) current and future uses; (ii) proximity and withdrawal rate of users; (iii) groundwater quantity and quality; (iv) the potential damage to wildlife, crops, vegetation, and physical structures caused by exposure to waste constituent; (v) the hydrogeologic characteristic of the facility and surrounding land; (vi) groundwater removal and treatment costs; and (vii) the cost and availability of alternative water supplies.
- Practicable capability of the owner or operator.
- Other relevant factors.

The time schedule for implementing and completing the remedial activity is influenced by many factors that should be considered by the owner or operator. The most critical factor is the nature and extent of the contamination, which significantly affects the ultimate treatment rate. The size of the treatment facility and the groundwater extraction and injection rates must be balanced for system optimization, capital resources, and remedial timeframe objectives. The nature of the contamination will influence the degree to which the aquifer must be flushed to remove adsorbed species. If contaminant migration is slow due to low transport properties of the aquifer, additional time may be available to evaluate the value of emerging and promising innovative technologies. The use of such technologies is not excluded as part of the requirement to implement a remedial action as soon as practicable. Delaying implementation to increase the availability of new technologies must be evaluated in terms of achievable cleanup levels, ultimate cost, additional environmental impact, and potential for increased risk to sensitive receptors. If a new technology clearly is superior to existing options in attaining remediation objectives, it may be appropriate to delay implementation. This may require that existing risks be controlled through interim measures. In setting the implementation schedule, the owner or operator should assess the risk to human health and the environment within the timeframe of reaching treatment objectives. If the risk is unacceptable, considering health-based assessments of exposure paths and exposure limits and the implementation time schedule must be accelerated or the selected remedy altered to provide an acceptable risk level in a timely manner.

Establishment of the schedule also may include consideration of the resource value of the aquifer, as it pertains to current and future use, proximity to users, quality and quantity of groundwater, agricultural value and uses (irrigation water source or impact on adjacent agricultural lands), and the availability of alternative supplies of water of similar quantity and quality. Based on these factors, a relative assessment of the aquifer's resource value to the local community can be established. Impacts to the resource and the degree of financial or health-related distress by users should be considered. The implementation timeframe should attempt to minimize the loss of value of the resource to users. The possibility that alternative water supplies will have to be developed as part of the remedial activities may need to be considered. Because owners or operators may not be knowledgeable in remediation activities, reliance on the owner or operator to devise the schedule for remediation may be impracticable. In these instances, the use of an outside firm to coordinate remediation scheduling may be necessary. Similarly, the development of a schedule for which the owner or operator cannot finance, when other options exist that do allow for owner or operator financing, should be prevented.

There are four situations where MSWLF units may not require the cleanup of hazardous constituents released to groundwater. If sufficient evidence exists to document that the groundwater is contaminated by a source other than the MSWLF unit, the director of an approved state may grant a waiver from implementing some or all of the corrective measure requirements. The owner or operator must demonstrate that the cleanup of a release from its MSWLF unit would provide no significant reduction in risk to receptors due to the concentrations of constituents from the other source. A waiver from corrective measures also may be granted if the contaminated groundwater is not a current or reasonably expected potential future drinking-water source, and it is unlikely that the hazardous constituents would migrate to waters causing an exceedance of GWPS. The owner or operator must demonstrate that the uppermost aquifer is not hydraulically connected with a lower aquifer. The owner or operator may seek an exemption if it can be demonstrated that attenuation, advection/dispersion, or other natural processes can remove the threat to interconnected aquifers. The owner or operator may seek the latter exemption if the contaminated zone is not a drinking-water resource. If remediation is not technically feasible or results in unacceptable cross-media impacts, then may waive cleanup requirements for any MSWLF units. Technical impracticabilities may be related to the accessibility of the groundwater to treatment, as well as the treatability of the groundwater using practicable treatment technologies. If the owner or operator can demonstrate that unacceptable cross-media impacts are uncontrollable under a given remedial option (i.e. movement in response to groundwater pumping or release of volatile organics to the atmosphere) and that the no action option is a less risky alternative, then the remediation is not necessary. A waiver of remedial obligation does not necessarily release the owner or operator from the responsibility of conducting source control measures or minimal groundwater remediation.

5.6.3.1. Monitoring activities

During the implementation period, groundwater monitoring must be conducted to demonstrate the effectiveness of the corrective action remedy. If the remedial action is not effectively curtailing further groundwater degradation or the spread of the contaminant plume, the replacement of the system with an alternative measure may be warranted. The improvement rate of the condition of the aquifer must be monitored and compared to the cleanup objectives. It may be necessary to install additional monitoring wells to more clearly evaluate remediation progress. In addition, if it becomes apparent that the GWPS will not be achievable technically, in a realistic time-frame, the performance objectives of the corrective measure must be reviewed and amended as necessary.

5.6.3.2. Interim measures

If unacceptable potential risks to human health and the environment exist prior to or during implementation of the corrective action, the owner or operator is required to take interim measures to protect receptors. These interim measures are typically short-term solutions to address immediate concerns and do not necessarily address long-term remediation objectives. Interim measures may include activities such as control of groundwater migration through high-volume withdrawal of groundwater or response to equipment failures that occur during remediation (e.g., leaking drums). If contamination migrates offsite, interim measures may include providing an alternative water supply for human, livestock, or irrigation needs. Interim measures also pertain to source control activities that may be implemented as part of the overall corrective action. This may include activities such as excavation of the source material or *in situ* treatment of the contaminated source. Interim measures should be developed with consideration given to maintaining conformity with the objectives of the final corrective action.

5.6.4. *Implementation*

An owner or operator is required to continue the assessment-monitoring program during the remedial action. Through monitoring, the short- and long-term successes of the remedial action can be gauged against expected progress. During the remedial action, it may be necessary to install additional groundwater monitoring wells or pumping or injection wells to adjust to conditions that vary from initial assessments of the groundwater flow system. As remediation progresses and data are compiled, it may become evident that the remediation activities will not protect human health and the environment, meet GWPSs, control sources of contamination, or comply with waste management standards. The reasons for unsatisfactory results may include:

- Refractory compounds that are not amenable to removal or destruction (detoxification).
- The presence of compounds that interfere with treatment methods identified for target compounds.
- Inappropriately applied technology.
- Failure of source control measures to achieve desired results.
- Failure of groundwater control systems to achieve adequate containment or the removal of contaminated groundwater.
- Residual concentrations above GWPSs that cannot be effectively reduced further because treatment efficiencies are too low.
- Transformation or degradation of target compounds to different forms that are not amenable to further treatment by present or alternative technologies.

The owner or operator should compare treatment assumptions with existing conditions to determine if assumptions adequately depict site conditions. If implementation occurred as designed, the owner or operator should attempt to modify or upgrade existing remedial technology to optimize performance and to improve treatment effectiveness. If the existing technology is found to be unable to meet remediation objectives, alternative approaches must be evaluated that could meet these objectives while the present remediation is continued. During this re-evaluation period, the owner or operator may suspend treatment only if the continuation of remedial activities clearly increases the threat to human health and the environment.

The regulatory period of compliance is three consecutive years at all points within the contaminant plume that lie beyond the groundwater monitoring system unless the specifies an alternative length of time for site-specific locations. Compliance is achieved when the concentrations of hazardous constituents do not exceed the GWPSs for a predetermined length of time. Upon the completion of the remedy, including meeting the GWPS at all points within the contaminant plume, the owner or operator must notify the concern authorities within 14 days that a certification that the remedy has been completed has been placed in the operating record. The certification must be signed by the owner or operator and a qualified groundwater scientist. Upon the completion of the remedial action, the owner or operator is released from the financial assurance requirements pertaining to corrective actions.

In determining an alternate period rather than three years the following issues must considered:

- The extent and concentration of the release(s).
- The behavior characteristics (fate and transport) of the hazardous constituents in the groundwater (i.e. mobility, persistence, toxicity, etc.).
- Accuracy of monitoring or modeling techniques, including any seasonal, meteorological or other.
- Environmental variabilities that may affect accuracy.
- The characteristics of the groundwater (i.e. flow rate, pH, etc.).

Consideration of these factors may result in an extension or shortening of the time required to show compliance with remediation objectives.

6. Closure and Post-Closure Care

6.1. *General Overview*

MSWLF closure focus on two central themes: (1) the need to establish low-maintenance cover systems and (2) the need to design a final cover that minimizes the infiltration of precipitation into the waste. Landfill closure technology, design, and maintenance procedures continue to evolve as geosynthetic materials become available, as performance requirements become more specific, and as limited performance history becomes available for the relatively small number of landfills that have been closed using current procedures and materials. Critical technical issues that must be faced by the designer include the:

- Degree and rate of post-closure settlement and stresses imposed on soil liner components;
- Long-term durability and survivability of cover system;
- Long-term waste decomposition and management of landfill leachate and gases; and
- Environmental performance of the combined bottom liner and final cover system.

6.2. *Final Cover Design*

Design criteria for a final cover system should be selected to:

- Minimize infiltration of precipitation into the waste;
- Promote good surface drainage;
- Resist erosion;
- Control landfill gas migration and/or enhance recovery;
- Separate waste from vectors (e.g., animals and insects);

- Improve esthetics;
- Minimize long-term maintenance;
- Protect human health and the environment; and
- Consider final use.

The first three points are directly related to the regulatory requirements. The other points typically are considered in designing cover systems for landfills.

Reduction of infiltration in a well-designed final cover system is achieved through good surface drainage and run-off with minimal erosion, transpiration of water by plants in the vegetative cover and root zone, and restriction of percolation through earthen material. The cover system should be designed to provide the desired level of long-term performance with minimal maintenance. Surface water run-off should be properly controlled to prevent excessive erosion and soil loss. Establishment of a healthy vegetative layer is key to protecting the cover from erosion. However, consideration also must be given to selecting plant species that are not deeply rooted because they could damage the underlying infiltration layer. In addition, the cover system should be geotechnically stable to prevent failure, such as sliding, that may occur between the erosion and infiltration layers, within these layers, or within the waste. Figure 36[41–43] illustrated the minimum requirements for the final cover system, liner system, and LCS for a typical MSWLF.

6.2.1. *Infiltration Layer*

The infiltration layer must be at least 18 in. (45–60 cm) thick and consist of earthen material that has a hydraulic conductivity less than or equal to the hydraulic conductivity of any bottom liner system or natural sub-soils. MSWLF units with poor or nonexistent bottom liners possessing hydraulic conductivities $>1 \times 10^{-5}$ cm/sec must have an infiltration layer that meets the 1×10^{-5} cm/sec minimum requirement. For units that have a composite liner with an FML, or naturally occurring soils with very low permeability (i.e. 1×10^{-7} cm/sec), anticipates that the infiltration layer in the final cover will include a synthetic membrane as part of the final cover. A final cover system for an MSWLF unit with an FML combined with a soil liner and LCS is presented in Fig. 36 and composite liner arrangement presented previously in Fig. 25.

The earthen material used for the infiltration layer should be free of rocks, clods, debris, cobbles, rubbish, and roots that may increase the hydraulic conductivity by promoting preferential flow paths. To facilitate run-off while minimizing erosion, the surface of the compacted soil should have a minimum slope of 3% and a maximum slope of 5% after allowance for settlement. It is critical that side slopes, which are frequently greater than 5%, be evaluated for erosion potential.

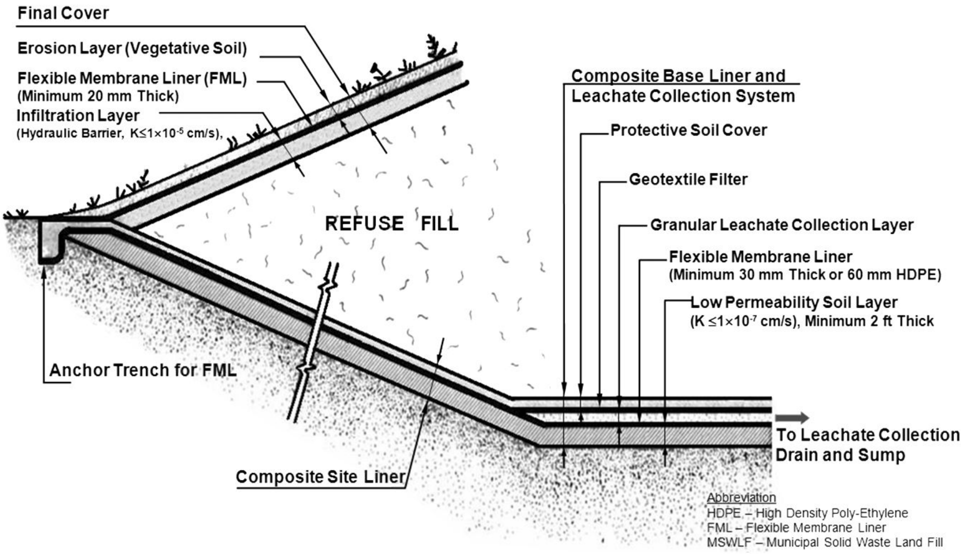


Figure 36. Final Cover, Liner, and LCS for a typical MSWLF.[41–43]

Membrane and clay layers should be placed below the maximum depth of frost penetration to avoid freeze-thaw effects. Freeze-thaw effects may include the development of microfractures or realignment of interstitial fines, which can increase the hydraulic conductivity of clays by more than an order of magnitude. Infiltration layers may be subject to desiccation, depending on climate and soil water retention in the erosion layer. Fracturing and volumetric shrinking of the clay due to water loss may increase the hydraulic conductivity of the infiltration layer. The infiltration layer is designed and constructed in a manner similar to that used for soil liners with the following differences:

- Because the cover is generally not subject to large overburden loads, the issue of compressive stresses is less critical unless post-closure land use will entail the construction of objects that exert large amounts of stress.
- The soil cover is subject to loadings from settlement of underlying materials. The extent of settlement anticipated should be evaluated and a closure and post-closure maintenance plan should be designed to compensate for the effects of settlement.
- Direct shear tests performed on construction materials should be conducted at lower shear stresses than those used for liner system designs.

If a geomembrane is used as an infiltration layer, the geomembrane should be at least 20 mils (0.5 mm) in thickness, although some geomembrane materials may need to be a greater thickness (i.e. a minimum thickness of 60 mils is recommended for HDPE because of the difficulties in making consistent field seams in thinner material). Increased thickness and tensile strengths may be necessary to prevent failure under stresses caused by construction and waste settlement during the post-closure care period. The strength, resistance to sliding, hydraulic performance, and actual thickness of geomembranes should be carefully evaluated. The quality and performance of some textured sheets may be difficult to evaluate due to the variability of the textured surface.

The design of a final cover is site-specific and the relative performance of cover design options may be compared and evaluated by the model. The HELP program calculates daily, average, and peak estimates of water movement across, into, through, and out of landfills. The input parameters for the model include soil properties, precipitation and other climatological data, vegetation type, and landfill design information. Default climatologic and soil data are available but should be verified as reasonable for the site modeled. Outputs from the model include precipitation, run-off, percolation through the base of each cover layer subprofile, evapotranspiration, and lateral drainage from each profile. The model also calculates the maximum head on the barrier soil layer of each subprofile and the maximum and minimum soil moisture contents of the evaporative zone. Data from the model are presented in a tabular report format and include the input parameters used and a summary of the simulation results. The HELP model may be used to estimate the hydraulic performance of the cover system designed for an MSWLF unit. Useful information provided by the HELP model includes surface run-off, duration, and quantity of water storage within the erosion layer, and net infiltration through the cover system to evaluate whether leachate will accumulate within the landfill.

6.2.2. *Erosion Layer*

The thickness of the erosion layer is influenced by depth of frost penetration and erosion potential. This layer is also used to support vegetation (Fig. 36). Erosion can adversely affect the performance of the final cover of an MSWLF unit by causing rills that require maintenance and repair. As previously stated, a healthy vegetative layer can protect the cover from erosion; conversely, severe erosion can affect the vegetative growth. Extreme erosion may lead to the exposure of the infiltration layer, initiate or contribute to sliding failures, or expose the waste. By evaluating erosion loss, the design may be optimized to reduce maintenance through selection of the best available soil materials or by initially adding excess soil to increase the time required before maintenance is needed. Parameters in the equation include the following:

$$X = \mathrm{RKLSCP}, \tag{8}$$

where X = soil loss (tons/acre/year), R = rainfall erosion index, K = soil erodibility index, L = slope length factor, S = slope gradient factor, C = crop/vegetation management factor, and P = erosion control practice.

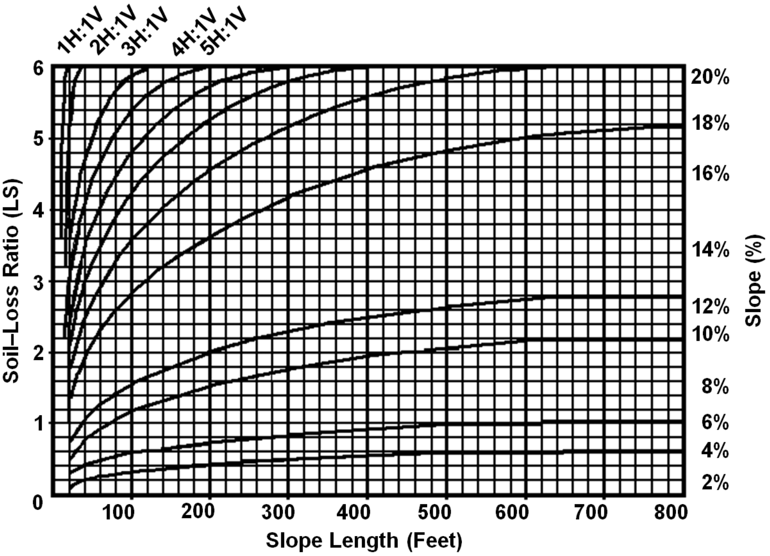


Figure 37. Chart of Soil Erosion Due to Slope.[41]

Values for the universal soil loss equation parameters may be obtained from local or regional SCS or land department. Figure 37[41] can be used to find the soil loss ratio due to the slope of the MSWL site. Loss from wind erosion can be determined by the following equation:

$$X' = I'K'C'L'V', \tag{9}$$

where X' = annual wind erosion, I' = field roughness factor, K' = soil erodibility index, C' = climate factor, L' = field length factor, and V' = vegetative cover factor.

A vegetative cover not only improves the appearance of the site, but also controls erosion of the final cover; a vegetated cover may require only minimal maintenance. The vegetation component of the erosion layer should have the following specifications and characteristics:

- Locally adapted perennial plants that are resistant to drought and temperature extremes;
- Roots that will not disrupt the low-permeability layer;
- The ability to thrive in low-nutrient soil with minimum nutrient addition;
- Sufficient plant density to minimize cover soil erosion;
- The ability to survive and function with little or no maintenance (i.e., self-supportive); and
- Sufficient variety of plant species to continue to achieve these characteristics and specifications over time.

The use of deep-rooted shrubs and trees is generally inappropriate because the root systems may penetrate the infiltration layer and create preferential pathways of percolation. Plant species with fibrous or branching root systems are suited for use at landfills, and can include a large variety of grasses, herbs (i.e., legumes), and shallow-rooted plants. The suitable species in a region will vary, dependent on climate and site-specific factors such as soil type and slope gradient and aspect. The timing of seeding (spring or fall in most climates) is critical to successful germination and establishment of the vegetative cover. Temporary winter covers may be grown from fast-growing seed stock such as winter rye.

The selection of the soil for the vegetative cover (erosion layer) should include the consideration of soil type, nutrient and pH levels, climate, species of the vegetation selected, mulching, and seeding time. Loamy soils with a sufficient organic content generally are preferred. The balance of clay, silt, and sand in loamy soils provides an environment conducive to seed germination and root growth. However, alternative design approach will be addressed in the areas that will not capable of sustaining plant growth.

6.3. *Alternative Final Cover*

An alternative material and/or an alternative thickness may be used for an infiltration layer as long as the infiltration layer requirements specified in as site-specific design criteria. For example, an armored surface (i.e. one composed of cobble-rich soils or soils rich in weathered rock fragments) could be used as an alternative to the 6-in. erosion layer. An armored surface, or hardened cap, is generally used in arid regions or on steep slopes where the establishment and maintenance of vegetation may be hindered by lack of soil or excessive run-off. The materials used for an armored surface typically are:

- Capable of protecting the underlying infiltration layer during extreme weather events of rainfall and/or wind;
- Capable of accommodating settlement of the underlying material without compromising the component;
- Designed with a surface slope that is approximately the same as the underlying soil (at least 2% slope); and
- Capable of controlling the rate of soil erosion.

The erosion layer may be made of asphalt or concrete. These materials promote run-off with negligible erosion. However, asphalt and concrete deteriorate due to thermal expansion and due to deformation caused by subsidence. Crushed rock may be spread over the landfill cover in the areas where weather conditions such as wind, heavy rain, or temperature extremes commonly cause the deterioration of vegetative covers.

6.3.1. *Additional Cover System Components*

To reduce the generation of post-closure leachate to the greatest extent possible, owners and operators can install a composite cover made of a geomembrane and a soil component with low-hydraulic conductivity. The hydraulic properties of these components are already discussed previously in design criteria section. Other components that may be used in the final cover system include a drainage layer, a gas vent layer, and a biotic barrier layer. These components are discussed in the following sections and are shown in Fig. 38.[41]

6.3.1.1. Drainage layer

A permeable drainage layer, constructed of soil or geosynthetic drainage material, may be constructed between the erosion layer and the underlying infiltration layer. The drainage layer in a final cover system removes percolating water that has infiltrated through the erosion layer after surface run-off and evapotranspiration losses. By removing water in contact with the low-permeability layer, the potential for leachate generation is diminished. Caution should be taken when using a drainage layer because this layer may prematurely draw moisture from the erosion layer that is needed to sustain vegetation. If a drainage layer is used, owners or operators should consider methods to minimize physical clogging of the drainage layer by root systems or soil particles. A filter layer, composed of either a low nutrient soil or geosynthetic material, may be placed between the drainage layer and the cover soil to help minimize clogging. If granular drainage layer material is used, the

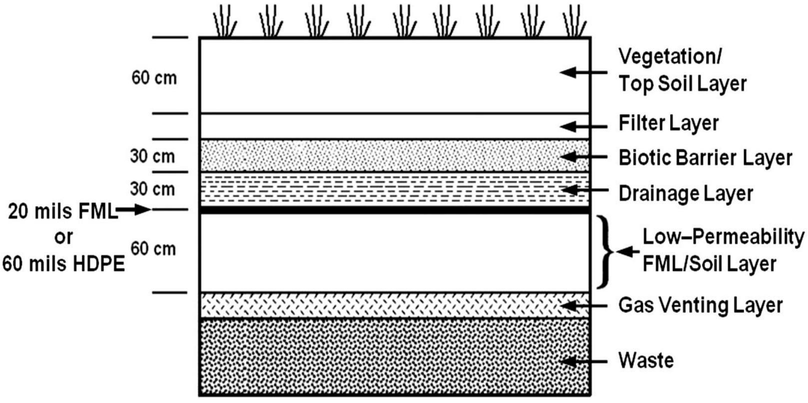


Figure 38. An Alternative Final Cover Design.[41]

filter layer should be at least 12 in. (30 cm) thick with a hydraulic conductivity in the range of 1×10^{-2} cm/sec to 1×10^{-3} cm/sec. The layer should be sloped at least 3% at the bottom of the layer. Greater thickness and/or slope may be necessary to provide sufficient drainage flow as determined by site-specific modeling. Granular drainage material will vary from site to site depending on the type of material that is locally available and economical to use. Typically, the material should be no coarser than 3/8 in. (0.95 cm), classified according to the universal soil classification system (USCS) as type SP, smooth and rounded, and free of debris that could damage an underlying geomembrane. Crushed stone generally is not appropriate because of the sharpness of the particles. If the available drainage material is of poor quality, it may be necessary to increase the thickness and/or slope of the drainage layer to maintain adequate drainage. The HELP model can be used as an analytical tool to evaluate the relative expected performance of alternative final cover designs. If geosynthetic materials are used as a drainage layer, the fully saturated effective transmissivity should be the equivalent of 12 in. of soil (30 cm) with a hydraulic conductivity range of 1×10^{-2} cm/sec to 1×10^{-3} cm/sec. Transmissivity can be calculated as the hydraulic conductivity multiplied by the drainage layer thickness. A filter layer (preferably a nonwoven needle punch fabric) should be placed above the geosynthetic material to minimize intrusion and clogging by roots or by soil material from the top layer.

6.3.1.2. Gas vent layer

Landfill gas collection systems serve to inhibit gas migration. The gas collection systems typically are installed directly beneath the infiltration layer. The function of a gas vent layer is to collect combustible gases (methane) and other potentially harmful gases (hydrogen sulfide) generated by microorganisms during biological decay of organic wastes, and to divert these gases via a pipe system through the infiltration layer. A detailed discussion concerning landfill gas, including the use of active and passive collection systems, is provided in previous section.

The gas vent layer is usually 12 in. (30 cm) thick and should be located between the infiltration layer and the waste layer. The materials used in construction of the gas vent layer should be medium- to coarse-grained porous materials such as those used in the drainage layer. Geosynthetic materials may be substituted for granular materials in the vent layer if equivalent performance can be demonstrated. Venting to an exterior collection point can be provided by means such as horizontal pipes patterned laterally throughout the gas vent layer, which channel gases to vertical risers or lateral headers. If vertical risers are used, their number should be minimized (as they are frequently vandalized) and located at high points in the cross section. Condensates will form within the gas collection pipes; therefore, the design should address the drainage of condensate to prevent blockage by its accumulation in low points. The most obvious potential problem with gas collection systems is the possibility of gas vent pipe penetrations through the cover system. Settlement within the landfill may cause concentrated stresses at the penetrations, which could result in infiltration layer or pipe failure. If a geomembrane is used in the infiltration layer, pipe sleeves, adequate flexibility, and slack material should be provided at these connections when appropriate. Alternatively, if an active gas control system is planned, penetrations may be carried out through the sides of the cover directly above the liner anchor trenches where the effects of settlement are less pronounced. The gas collection system also may be connected to the LCS, both to vent gases that may form inside the leachate collection pipes and to remove gas condensates that form within the gas collection pipes. This method generally is not preferred because if the leachate collection pipe is full, gas will not be able to move through the system.

6.3.1.3. Biotic layer

Deep plant roots or burrowing animals (collectively called biointruders) may disrupt the drainage and the low-hydraulic conductivity layers, thereby interfering with the drainage capability of the layers. A 30-cm (12-in.) biotic barrier of cobbles directly beneath the erosion layer may stop the penetration of some deep-rooted plants and the invasion of burrowing animals. Most researches on biotic barriers have been done in, and is applicable to arid areas. Geosynthetic products that incorporate a time-released herbicide into the matrix or on the surface of the polymer also may be used to retard plant roots. The longevity of these products requires evaluation if the cover system is to serve for longer than 30–50 years.

6.3.2. *Settlement and Subsidence*

Excessive settlement and subsidence, caused by the decomposition and consolidation of the wastes, can impair the integrity of the final cover system. Specifically, settlement can contribute to:

- Ponding of surface water on the cap;
- Disruption of gas collection pipe systems;
- Fracturing of low-permeability infiltration layers; and
- Failure of geomembranes.

The degree and rate of waste settlement are difficult to estimate. Good records regarding the type, quantity, and location of waste materials disposed will improve the estimate. Settlement due to consolidation may be minimized by compacting the waste during daily operation of the landfill unit or by landfilling baled waste. Organic wastes will continue to degrade and deteriorate after closure of the landfill unit.

Several models have been developed to analyze the process of differential settlement. Most models equate the layered cover to a beam or column undergoing deflection due to various loading conditions. If the amount of total settlement can be estimated, either from an analytical approach or from the empirical relationships from data collected during the operating life of the facility, the designer should attempt to estimate the potential strain imposed on the cover system components. Due to the uncertainties inherent in the settlement analysis, a biaxial strain calculation should be sufficient to estimate the stresses that may be imposed on the cover system. The amount of strain that a liner is capable of enduring may be as low as several percentage; for geomembranes, it may be 5–12%. Geomembrane testing may be included as part of the design process to estimate safety factors against cover system failure.

The cover system may be designed with a greater thickness and/or slope to compensate for settlement after closure. However, even if settlement and subsidence are considered in the design of the final cover, ponding may still occur after closure and can be corrected during post-closure maintenance. The cost estimate for post-closure maintenance should include earthwork required to regrade the final cover due to total and differential settlements. Based on the estimates of total and differential settlements from the modeling methods described earlier, it may be appropriate to assume that a certain percentage of the total area needs regrading and then incorporate the costs into the overall post-closure maintenance cost estimate.

6.3.3. *Sliding Instability*

The slope angle, slope length, and overlying soil load limit the stability of component interfaces (geomembrane with soil, geotextile, and geotextile/soil). Soil water pore pressures developed along interfaces also can dramatically reduce stability. If the design slope is steeper than the effective friction angles between the materials, sliding instability generally will occur. Sudden sliding has the potential to cause tears in geomembranes, which require considerable time and expense to repair. Unstable slopes may require remedial measures to improve stability as a means of offsetting potential long-term maintenance costs.

The friction angles between various media are best determined by laboratory direct shear tests that represent the design loading conditions. Methods to improve stability include using designs with flatter slopes, using textured material, constructing benches in the cover system, or reinforcing the cover soil above the membrane with geogrid or geotextile to minimize the driving force on the interface of concern.

6.4. *Closure Plan and Criteria*

The closure plan is a critical document that describes the steps that an owner or operator will take to ensure that all units will be closed in a manner that is protective of human health and the environment. The portions of the landfill unit that have not received a final cover must be included in the estimate. The area to be covered at any point during the active life of the operating unit can be determined by examining design and planned operation procedures and by comparing the procedures with construction records, operation records, and field observations. Units are operated frequently in phases, with some phases conducted on the top of previously deposited waste. If the owner or operator routinely closes landfill cells as they are filled, the plan should indicate the greatest number of cells open at one time. The estimate must account for the maximum amount of waste on-site that may need to be disposed in the MSWLF unit over the life of the facility (this includes any waste on-site yet to be disposed). The maximum volume of waste ever on-site can be estimated from the maximum capacity of each unit and any operational procedures that may involve the transfer of wastes to off-site facilities. Where insufficient design, construction, and operational records are found, areas and volumes may be estimated from topographic maps and/or aerial photographs. Steps that may be included in the closure plan are as follows:

- Notifying to concern authorities to initiate closure;
- Determining the area to receive final cover;
- Developing the closure schedule;
- Preparing construction contract documents and securing a contractor;
- Hiring an independent registered professional engineer's to observe closure activities and provide certification;
- Securing borrow material;
- Constructing the cover system;
- Obtaining signed certificate and placing it in operating record;
- Notifying State Director that certificate was placed in operating record; and
- Recording notation in deed to land or other similar instrument.

The closure plan should include a description of the final cover system and the methods and procedures that will be used to install the cover. The description of the methods, procedures, and processes may include design documents; construction specifications for the final cover system, including erosion control measures; QC testing procedures for the construction materials; and QA procedures for construction. Closure activities must begin within 30 days of the last receipt of waste and must be completed within 180 days. Some MSWLF units, such as those in seasonal population areas, may have remaining capacity but will not receive the next load of waste for a lengthy period. These MSWLF units must receive waste within one-year or they must close. Extensions to both the one year and the 180-day requirements may be available to owners or operators of MSWLF units. An extension may be granted if the owner or operator can demonstrate that there is remaining capacity or that additional time is needed to complete closure. These extensions could be granted to allow leachate recirculation or to allow for settlement. The owner or operator must take, and continue to take, all steps necessary to prevent threats to human health and the environment from the unclosed MSWLF unit. In general, this requirement should be established for a unit in compliance with the requirements of. The owner or operator may need to demonstrate how access to the unclosed unit will be controlled prior to closure or receipt of waste and how the various environmental control and monitoring systems (e.g., surface run-off, surface run-on, leachate collection, gas control system, and groundwater and gas monitoring) will be operated and maintained while the unit remains unclosed. Following closure of each MSWLF unit, the owner or operator must have a certification, signed by an independent registered professional engineer, verifying closure. The certificate should verify that closure was completed in accordance with the closure plan. This certification should be based on knowledge of the closure plan, observations made during closure, and documentation of closure activities provided by the owner or operator. The signed certification must be placed in the operating record and must be notified that the certification was completed and placed in the record. After closure of all units at an MSWLF facility, the owner or operator must record a notation in the deed, or in records typically examined during a title search, that the property was used as an MSWLF unit and that its use is restricted for general purpose.

6.5. *Post-closure Care and Plan*

When the final cover is installed, repairs and maintenance may be necessary to keep the cover in good working order. Maintenance may include the inspection, testing, and cleaning of leachate collection and removal system pipes, the repairs of final cover, and the repairs of gas and groundwater monitoring networks. Inspections should be made on a routine basis. A schedule should be developed to check that routine inspections are completed. Records of inspections detailing observations should be kept in a log book so that changes in any of the MSWLF units can be monitored; in addition, records should be kept detailing changes in post-closure care personnel to ensure that changing personnel will not affect post-closure care due to the lack of knowledge of routine activities. The activities and frequency of inspections are subject to concern authorities review to ensure that units are monitored and maintained for as long as is necessary to protect human health and the environment. Inspection of the final cover may be performed on the ground and through aerial photography. Inspections should be conducted at appropriate intervals and the condition of the facility should be recorded with notes, maps, and photographs. The inspector should take notice of eroded banks, patches of dead vegetation, animal burrows, subsidence, and cracks along the cover. The inspector also should note the condition of concrete structures (i.e. manholes), leachate collection and removal pipes, gas monitoring systems, and monitoring wells. For larger facilities, annual aerial photography or high resolution satellite imagery observation may be a useful way to document the extent of vegetative stress and settlement if either of these has been observed during routine inspections. It is important to coordinate the photography with the site "walkover" to verify interpretations made from aerial photographs. Aerial photography should not be used in place of a site walkover but in conjunction with the site walkover. Topographic surveys of the landfill unit(s) may be used to determine whether or not settlement has occurred. These should be repeated every few years until settlement behavior is established. If settlement plates are used, they should be permanent and protected from vandalism and accidental disturbance. Depressions caused by settlement may lead to ponding and should be filled with soil. Excessive settlement may warrant reconstructing or adding to portions of the infiltration layer. Damage caused by settlement such as tension cracks and tears in the synthetic membrane should be repaired.

Cover systems that have areas where the slope is greater than 5% may be susceptible to erosion. Large and small rills (crevices) may form along the cover where water has eroded the cover. This may lead to exposure of the synthetic geomembrane and, in severe cases, depending on the cover system installed, exposure of the waste. Erosion may lead to increased infiltration of surface water into the landfill. Areas showing the signs of erosion should be repaired. Certain types of vegetative cover (i.e. turf-type grasses) may require mowing at least two times a year. Mowing can aid in the suppression of weed and brush growth, and can increase the vigor of certain grass species. Alternatively, certain cover types (i.e. native prairie grasses) require less frequent mowing (once every three years) and may be suitable for certain climates and facilities where a low-maintenance regime is preferable. For certain cover types, fertilization schedules may be necessary to sustain desirable vegetative growth. Fertilization schedules should be based on the cover type present. Annual or biennial fertilization may be necessary for certain grasses, while legumes and native vegetation may require little or no fertilizer once established. Insecticides may be used to eliminate insect populations that are detrimental to vegetation. Insecticides should be carefully selected and applied with consideration for potential effects on surface water quality.

Some leachate collection and removal systems have been designed to allow for inspections in an effort to ensure that they are working properly. Leachate collection and removal pipes may be flushed and pressure-cleaned on regularly (i.e. annually) to reduce the accumulation of sediment and precipitation and to prevent biological fouling. Similarly, gas collection systems should be inspected to ensure that they are working properly. Vents should be checked to ensure they are not clogged by foreign matter such as rocks. If not working properly, the gas collection systems should be flushed and pressure-cleaned. At some landfill facilities, leachate concentrations eventually may become low enough so as not to pose a threat to human health or the environment. However, owner or operator allows to cease managing leachate if he or she can demonstrate that the leachate no longer poses a threat to human health and the environment. The demonstration should address direct exposures of leachate releases to groundwater, surface water, or seeps. Indirect effects, such as accumulated leachate adversely affecting the chemical, physical, and structural containment systems that prevent leachate release, also should be addressed in the demonstration. The threat posed by direct exposures to leachate released to groundwater, to surface waters, or through seeps may be assessed using health-based criteria. These criteria and methods are available through the integrated risk information system (IRIS), including MCLs established under the safe drinking-water regulations and the ambient water quality criteria under the CWA. Concentrations at the points of exposure, rather than concentrations in the leachate in the collection system, may be used when assessing threats.

The post-closure plan should provide a schedule for routine maintenance of the MSWLF unit systems. These systems include the final cover system, the leachate collection and removal system, and the landfill gas and groundwater monitoring systems. The plan must include the name, address, and telephone number of the person or office to contact regarding the facility throughout the post-closure period. Additionally, the planned uses of the property during the post-closure period must be provided in the plan. These uses may not disturb the integrity of the final cover system, the liner system, and any other components of the containment or monitoring systems. Any other disturbances to any of the MSWLF components must be approved by the concern authorities.

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Chapter 4

LANDFILLING OF MUNICIPAL SOLID WASTE IN EUROPE

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Abstract
Under the hierarchy of waste management, landfilling is the least desirable option, but it is still commonly practiced. Although biodegradation processes ensure neutralization and stabilization of waste, but because of the long-term period of waste degradation, landfills pose threat to the environment. In order to minimize detrimental influence on the surroundings, landfills should be properly constructed and managed for all the operation time and after the closure. The chapter presents the standards for landfill construction and operation binding in the European Union.

Keywords: Municipal solid waste, landfilling, capping systems, liner systems, landfill gas, leachate, monitoring.

1. Waste Landfilling — Current Status and Perspectives

1.1. *Hierarchy of Waste Management*

The European Union (EU) strategy on waste has developed the concept of the "hierarchy of waste management."[1] The hierarchy originated in 1975 when Waste Framework Directive encouraged waste reduction, reuse, and recovery. The hierarchy was formally adopted in 1989 in the EU Community Strategy for Waste Management.[2] Under the applicable hierarchy of waste management, deposition of wastes on landfill is the least desirable option. Although biodegradation processes ensure that biodegradable waste is neutralized and stabilized to inert material, but throughout the long-term degradation period, landfills pose serious threat to the environment. Ideally, landfill should be repository exclusively for inert materials. Successful application of this concept requires extensive mechanical-biological (mechanical sorting as a first step) or biological-mechanical (biodegradation as a first step) waste pretreatment. Treatment is defined as physical, chemical, or biological processing, including sorting, that changes the characteristics of the waste in order to reduce its volume or hazardous nature, to facilitate its handling or enhance recovery.

The main idea of the modern waste deposition is landfilling sustainability. The landfills should be remaining in equilibrium with the environment preferably within the timeframe of one generation (30–50 years).

According to Kavazanjian,[3] three elements of sustainable landfill practices are:

1. protection of the human health and the environment,
2. maximizing the capacity of the existing landfills and minimizing the need for new landfills, and
3. making beneficial use of closed landfill sites.

Constructing landfills in a sustainable way limits a threat to human health or the environment because pollutants are broken down to harmless substances and/or flushed out (and rendered harmless) or immobilized in the landfill and consequently are not released.

1.2. *Legal Framework of Waste Landfilling in the European Union*

The management of waste via landfill is regulated throughout the European Union by the EC Waste Landfill Directive (Council Directive 1999/31/EC).[4] One of the aims of the Directive was to harmonize standards throughout the European Union for waste landfill facilities providing high level of environmental protection. European Commission regards landfilling as the least favorable option of waste management which does not make use of waste as a resource and may result in negative influence on the environment.

The Directive sets up a system of operating permits for landfill sites. Applications for permits must contain the following information:

- the identity of the applicant and, in some cases, of the operator;
- a description of the types and total quantity of waste to be deposited;
- the capacity of the disposal site;
- description of the site;
- proposed methods for pollution prevention and abatement;
- proposed operation, monitoring, and control plan;
- plan for closure and aftercare procedures;
- applicant's financial security; and
- environmental impact assessment study, required under Council Directive 85/337/EEC[5] on the Assessment of the Effects of Certain Public and Private Projects on the Environment.

The European Community has a strategy to reduce the emission of greenhouse gases; therefore, the Waste Landfill Directive[4] sets target to reduce the amount of biodegradable waste sent to landfill to 75% of the 1995 levels by 2006, 50% by 2009, and 35% by 2016. Countries that landfilled more than 80% of their municipal solid waste (MSW) in 1995 are allowed to extend the deadlines by four years (first stage by 2010, second by 2013, and third by 2020). The Directive introduced stringent operational and technical requirements for landfill, which are described further.

European Waste Catalogue (Commission Decision 2000/532/EC 2000)[6] contains categories or chapters of waste each with a two-digit code, each chapter contains sub-chapters and each sub-chapter lists the specific waste categories. Consequently, each category of waste has a specific six-digit code.

The UK Environmental Agency has issued series of documents that support the overarching document guidance on the management of landfill gas (2004).[7]

The full series comprises:

- Guidance for monitoring trace components in landfill gas,
- Guidance on landfill gas flaring,
- Guidance for monitoring enclosed landfill gas flares,
- Guidance for monitoring landfill gas engines emission,
- Guidance for monitoring landfill gas surface emission, and
- Guidance on treatment technologies for landfill gas engines.

The Agency has also issued the guidance for the treatment of landfill leachate and guidance on monitoring of landfill leachate, groundwater, and surface water.

1.3. *Strategies of Waste Landfilling*

Different tendencies of a modern landfilling can be indicated:

- Landfill as a final inert waste deposit site that does not influence the environment significantly (dominating in the European Union),
- Landfill as an anaerobic bioreactor (developing in the United States and Canada), which is used for intensive landfill gas production,
- Landfill as an aerobic bioreactor (developing in the European Union),
- Entombment (dry) landfill, containing waste in relatively dry form in order to prevent biodegradation (common in the USA and France (Kjeldsen *et al.*[8] after Williams[1])).

Leachate recirculation landfill technology (LRLT) is a kind of bioreactor landfill technology. It differs from bioreactor landfill technology only by the source of the recirculated liquid. Typical bioreactors add additional liquid, other than leachate or gas condensate, to bring the waste to a moisture content of 40%. Leachate recirculation landfills use only recirculated leachate (and gas condensate) produced at the landfill. This tends to limit the ability to achieve a waste moisture content of 40%. The processes taking place in LRLT are the same as in typical bioreactors and traditional landfills.

Potential advantages of this technology are:

- enhancement of gas production — a high moisture content and the movement of moisture promote methanogenesis,
- increase of landfill space,
- increase of the rate of waste settlement,
- improvement of leachate management — by the increase of its uniform quality (measured as COD), so that the design and operation of treatment and disposal facilities are easier,

- acceleration of hazardous organic compounds removal, by optimizing conditions for biodegradation,
- minimization of dry zones in the waste, improving the biodegradation conditions and reducing the time of landfill bioactivity,
- enhancement of evaporative losses of leachate by surface spraying,
- acceleration of waste stabilization, and
- reduction of post-close time and cost.

Landfills with leachate recirculation are becoming more common. It is more developed in North America than the bioreactor landfill technology.[9]

According to Reinhart *et al.*,[10] disadvantages and potential problems connected with LRLT are as follows:

- surface flooding — which may be caused either by uneven distribution of irrigation intensity or by water permeability,
- spray drift from leachate recirculation may result in health concern and increase of odors intensity,
- break-outs of leachate accumulated as perched water from the side slopes of landfills may occur, increased by the presence of the compacted or low-permeability layers within the waste,
- clogging of sub-surface recirculation system may occur, and
- extremely high concentration of dissolved salts may occur in sites accepting predominantly inorganic waste.

Other type of landfill represents the aerobic bioreactor. The aerobic bioreactor seeks to accelerate waste degradation by optimizing conditions for aerobes. Aerobes require more water to function compared to anaerobes. However, aerobic organisms can grow more quickly than anaerobes, because aerobic respiration is more efficient at generating energy. The aerobic degradation of waste can proceed faster than anaerobic one that shortens the landfill activity time. Another consequence is that aerobic respiration can generate large amount of metabolic heat, which requires significant quantities of water. Landfill aerobic activity is promoted by injection of air or oxygen into the waste mass. It is also possible to apply vacuum to the waste mass and pull air in through a permeable cap. Liquids are usually added through leachate recirculation. The aerobic process does not generate methane.[9]

ECS (Environmental Control Systems) aerobic landfill technology "uses a patented process that stabilizes waste far faster than traditional anaerobic methods — thus minimizing risk, enhancing revenues and extending the life of the facility, rather than waiting for the organic compounds to degrade naturally, which can take up to 30 years. ECS treats the material by injecting air and moisture directly into the waste. Degradation takes place in just one to three years and produces a nontoxic material that can be mined, recovered, or reused. That, in turn, can free up an additional 80–90% of airspace, which eliminates the need for new permits and expansions and ensures a longer-lasting facility that generates more profits”.[11]

The environmental and economic benefits of this technology are as follows:[11]

- Elimination of methane production, showing results within four weeks.
- Elimination of leachate treatment and disposal — in more than 95% of landfills.
- Elimination of VOCs, as well as greenhouse gases and odors.
- Elimination of groundwater contamination and the need and cost of monitoring.
- Round-the-clock monitoring from anyplace in the world.
- Elimination or reduction of closure costs because the facility lasts longer and protects the environment better.
- Enabling of reclamation of land for uses other than landfilling.

Implementation of landfill bioreactor technology (anaerobic or aerobic) requires modification of design and operational criteria associated with traditional landfilling. The impenetrable barriers have to be provided and waste stabilization must be enhanced and accelerated. The bottom liner system has to be designed to accommodate the additional flows contributed by leachate recirculation. The gas management facilities have to be operated to control amplified gas production. Overcompaction of waste during placement may adversely impact leachate routing and prevent even moisture distribution; due to this, the use of permeable alternative daily cover is recommended. Monitoring of leachate and gas quality and quantity becomes critical to operational decision making. Shredding and screening of waste are desirable to promote biological and chemical processes. Leachate management must be carefully planned.

1.4. *Environmental Impact of Landfills*

The land disposal has a long tradition, dating back to prehistoric times but problem began as society developed and population density increased. Land disposal of waste was initially not organized. Land depressions and artificial excavations were filled by waste. Complete lack of bottom and capping barriers was a typical feature of the open dumps. The dumps started to create environmental and human health problems that prompted innovations in design and operation. The sanitary landfill developed when special construction, controlled operation, and disposal techniques such as daily cover and compaction were found.

Despite of significant landfill technology progresses, a large number of adverse impacts occur from landfill operations. The impact can be caused by:

- landfill leachate causing pollution of groundwater and surface water,

- landfill biogas can be a reason of air pollution (emission of the greenhouse gases — methane, carbon dioxide and emission of malodorous and toxic gases) and creation of explosive conditions,
- noise and vibrations generated from vehicles accessing a landfill as well as from working face operations,
- aerosols and bio-aerosols released from the landfill surface,
- animal (birds and rats), insect (flies), and vermin vectors of disease, and
- mechanical pollution connected with fine and light waste spreading.

These impacts can vary from fatal accidents (e.g., scavengers buried under waste piles) and infrastructure damage (e.g., damage to access roads by heavy vehicles) to the pollution of the local environment (such as contamination of groundwater and/or aquifers by leakage and residual soil contamination during landfill usage, as well as after landfill closure) and to cause the health threats.

Despite various disadvantages, landfilling is still the most dominant disposal route for MSW in the European Union. For many countries, including United Kingdom, Italy, Spain, Finland, Portugal, Italy, Ireland, Turkey, Romania, Poland, Hungary, Bulgaria, Lithuania, and Latvia, more than 60% of MSW was disposed of in landfills (data on 2003).[12]

2. Chemical and Physical Processes in Landfill Body

MSW, constituting a majority of waste deposited in nonhazardous landfills, contains a high proportion of organic material (usually more than 50%), which can be successfully degraded by microorganisms. The process of waste degradation involves also physical and chemical processes.

2.1. *Chemical Evolution of Landfilled Waste*

Five main stages of degradation process in waste body have been identified[1,8,13]:

Stage I. Hydrolysis/aerobic degradation — It occurs during the emplacement of the waste and for a period thereafter. The aerobic microorganisms metabolize a portion of the organic fraction of waste to produce simpler hydrocarbons, carbon dioxide, water, and heat. The temperature rises to 70–90°C. Water absorbs the carbon dioxide, which gives acidity to the leachate.

Stage II. Hydrolysis and fermentation — After depletion of available oxygen, different microorganisms able to tolerate anoxic conditions become dominant. Products of the first stage processes, such as carbohydrates, protein, and lipids are hydrolyzed to sugars and further to carbon dioxide, hydrogen, ammonia, and organic acids. Proteins are being decomposed to ammonia, carboxylic acids, and carbon dioxide. Temperature drops to 30–50°C.

Stage III. Acetogenesis — Acetogen microorganisms convert the organic acids into acetic acid and its derivatives, carbon dioxide and hydrogen, under anaerobic conditions. Other kinds of microorganisms convert carbohydrates directly to acetic acid in the presence of carbon dioxide and hydrogen. Low hydrogen concentrations promote the methane-producing bacteria (methanogens), which generate methane and carbon dioxide from organic acids and their derivatives. The pH value is low; therefore, the solubility of metal ions increases.

Stage IV. Methanogenesis — It is the main landfill gas generation stage. The gas composition is about 60% of methane and 40% of carbon dioxide. Low level of hydrogen is necessary to promote methanogens to produce methane from organic acids. Methane can be also produced directly from hydrogen and carbon dioxide. Two types of bacteria are active in this phase: mesophilic bacteria (temperatures 30–35°C) and thermophilic bacteria (temperatures 45–60°C). The pH value rises to about 7–8. It is the longest stage of waste degradation.

Stage V. Oxidation — End of the degradation reaction causes the new aerobic stage, which is called oxidation. Methanotrophic bacteria, which convert residual methane to carbon dioxide and water, appear.

Changes in the landfill gas and leachate compositions occurring throughout waste degradation process are presented in Fig. 1. Because of the heterogeneous nature of waste, all the different stages may be progressing simultaneously until the stabilization of the landfill has been reached.

The main factors influencing the waste biodegradation process are:[1]

- Site characteristics — Waste layer depths exceeding 5 m tend to develop anaerobic conditions and greater quantities of landfill gas; capping system reduces the air and rainfall penetration to waste.
- Waste characteristics — Quantity and quality of biodegradable fraction influence the rate of waste decomposition. Particle size influences the conditions of gas penetration in waste layer and microorganisms accessibility to waste. High waste density resulted from the compaction may limit the percolation of water through the landfill.
- Moisture content — It influences the growth of bacteria. It depends on inherent moisture content of waste and the rainfall level in the area. Usually, it ranges from

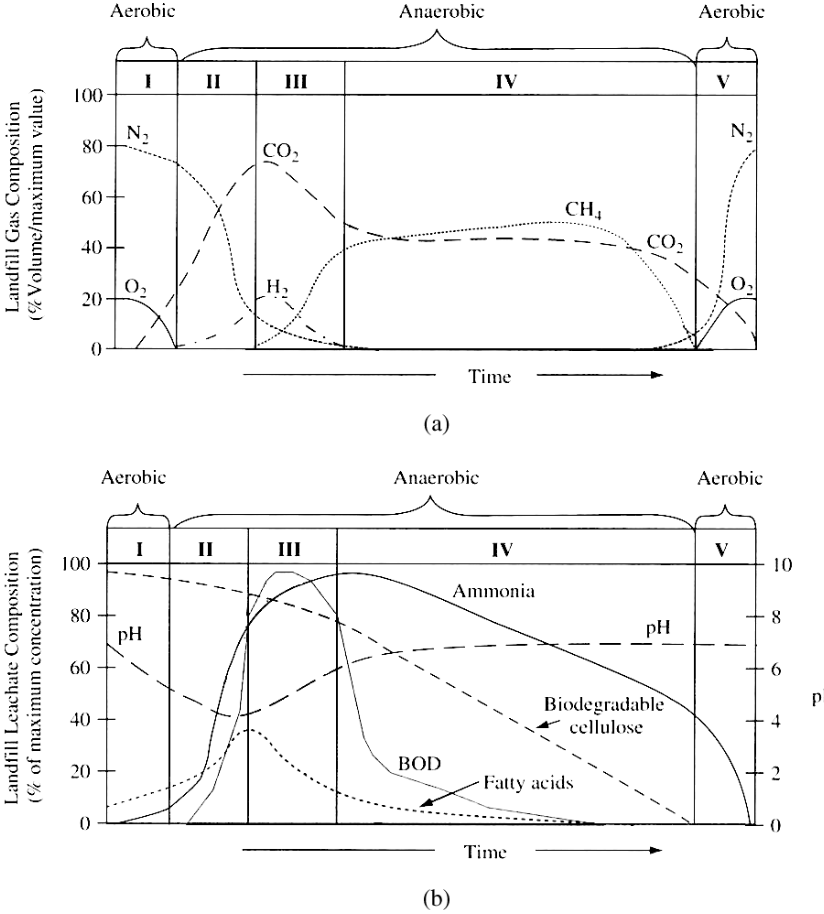

Figure 1. Landfill Gas Composition and Leachate Composition in Relation to the Degradation of Biodegradable Solid Waste (Waste Management Paper,[14] after Williams,[1] Modified).

15% to 40%. Moisture content can be most practically controlled by leachate recirculation.

- pH — pH value changes according to the landfill stage; initially it is neutral, next it is followed by acidic phase (decreases to 4), and finally it rises to the range 6.8–7.5.
- Temperature — Exothermic processes in first stage increase the temperature to 80°C. Compacted wastes reach lower temperatures due to lower availability of oxygen. The temperatures in anaerobic stages drop to 30–40°C. The cold climate reduces the rate of landfill gas production; it was found that mesophilic bacteria dominate in 1–5 m depths and thermophilic bacteria in the deeper parts of landfill.

- Nutrient availability and absence of toxins for bacteria — Nutrient availability affects microbial activity and both deficiency and surplus of nutrients can occur locally or generally in the landfill. The same is valid for presence of toxins for bacteria.
- Oxidation–reduction potential — Reflects electron availability for redox reactions, i.e. for reactions with transfer of electrons. It decreases from initial values that may be as high as $+800$ mV in the initial aerobic phase, to as low as -400 mV during hydrogen production phase increasing again in the phase V of return of aerobic conditions. Methane is usually produced at redox potentials below -200 mV.

2.2. *Mechanical Properties and Behavior of Landfills*

Mechanical properties of waste body such as waste bulk density, porosity, water holding capacity, water content, and hydraulic conductivity/permeability change with time during the degradation process, mainly due the mechanical compaction but also due to natural settlement resulting from gravity and mineralization of the waste organic fraction. Mechanical properties are influenced also by the waste pretreatment process and type of waste stabilization.

A decisive criterion for the construction and operation of landfill is the emplacement density of dry waste that can be achieved in practice. High emplacement densities lead to greater tension at the landfill base and the base liner and leachate drainage system must be dimensioned for higher loads. According to the examination carried out by Bauer *et al.*,[15] the Proctor densities for aerobically stabilized waste were in the range of 0.70–1.2 Mg/m^3, whereas materials from anaerobic treatment are tending to achieve smaller Proctor densities. Mechanical-biological treatment causes a clear increase of the attainable bulk density of the waste as well as a change of the strength behavior.

The higher density and the smaller pore volume lead to a lower hydraulic conductivity. The hydraulic conductivity determines the water infiltration into the waste body.[16] The correlation of density and vertical permeability of waste is shown in Fig. 2.

The settlement behavior of MSW deposited in landfill body is often classified into five phases[17–19]:

1. physical compression and creep due to mechanical distortion, bending, crushing, and reorientation,
2. raveling settlement due to the migration of small particles into voids among large particles,
3. viscous behavior and consolidation phenomena involving both solid skeleton and single particles or compounds,

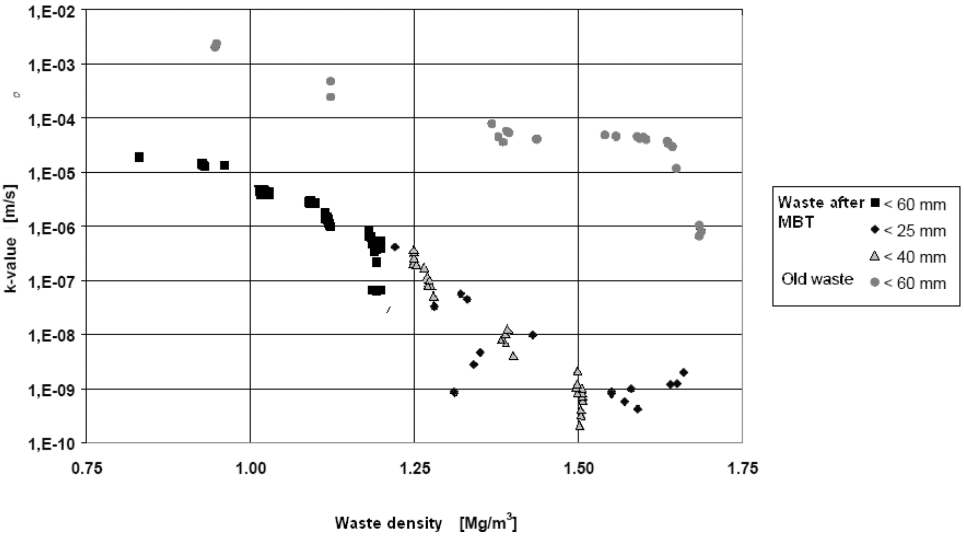

Figure 2. Influence of the Density on the Vertical Permeability for Waste of Different Particle Size (Münnich *et al.*,[16] Modified).

4. decomposition settlement due to the biodegradation of the organic compounds, and
5. collapse of components due to physicochemical changes such as corrosion, oxidation, and degradation of inorganic compounds (residual deformation).

Total settlement of the untreated waste deposited in the landfill ranges from 40% to 20% of the initial waste layer height. The compression test that had been carried out in large-scale pedometers with fresh waste showed that the total settlement was approximately 35% of the initial height.[15] The examination carried out by Bowders *et al.*[19] shown that wet cell exhibited 1.5–4.5 times more settlement than the dry cell. According to EU Landfill Directive,[4] the level of waste settlement due to its consolidation process during degradation stages should be evaluated.

3. Landfill Design

3.1. *Selection of Landfill Type*

The selection of landfill types depends on the waste quality. According to EC Waste Directive,[4] waste landfills are categorized into three groups:

1. waste landfills for hazardous waste,
2. waste landfills for nonhazardous waste, and
3. waste landfills for inert waste.

The list of waste acceptable at each designated and permitted landfill site is a part of the permitting process of the regulatory authorities for each country. General principles of acceptance are composition, leachability, long-term behavior, and general properties. The acceptance criteria for landfill would be based on the standardized waste analysis and limit values for particular features.

Nonhazardous waste includes MSW and a wide range of industrial waste, such as organic and inorganic wastes. The nonhazardous wastes are usually "bioreactive." It allows using landfill as a bioreactor. In this type of landfill, the biodegradation of waste occurs under proper conditions that increase the rate of MSW decomposition. Additionally, decrease of environmental risk, increase of gas production rate, shorter duration of landfill impact on environment, improvement of leachate quality, and more rapid landfill settlement are observed.

Nonreactive hazardous wastes, for example, those that are solidified or vitrified, are also permitted to be deposited into nonhazardous waste landfills, but there is a requirement that such waste should be placed in special cells that do not contain biodegradable wastes.

Some types of hazardous wastes, e.g., liquid waste, flammable waste, explosive waste or oxidizing waste, infectious clinical or hospital waste, and used tires, are not permitted to go to landfill sites. Inert wastes (bricks, glass, tiles, ceramic materials, and stones) do not undergo any significant physical, chemical, or biological transformations and they are deemed not to pose a significant environmental reactivity.

According to the localization of waste mass in relation to land surface level, landfills can be grouped into:

- underground landfills,
- above-ground landfills, and
- in-ground landfills.

3.2. *Selection of Landfill Location*

The Waste Landfill Directive[4] sets out the location requirements on which the landfill operator has to report, to the competent regulatory authority, to ensure that the proposed landfill does not pose a serious environmental risk.

The location of the landfill must take into consideration requirements relating to:

(a) the distances from the boundary of the site to residential and recreational areas, waterways, water bodies, and other agricultural or urban sites;
(b) the existence of groundwater, coastal water, or nature protection zones in the area;

(c) the geological and hydrogeological conditions in the area (kind of bedrock, mineralogical and grain size composition, permeability of the ground, and groundwater level);
(d) the risk of flooding, subsidence, landslides, or avalanches on the site; and
(e) the protection of the nature or cultural patrimony in the area.

The meteorological characteristics of area and the availability of final and daily covering materials should be assessed. As it was already mentioned, for the large landfills, an environmental impact assessment is required.

According to EU regulation requirements, the bottom of landfill must be located at least 1 m above the groundwater level.

4. Sanitary Landfill Construction

4.1. *Components of Sanitary Landfill*

Main components of sanitary landfill are as follows (Fig. 3):

Infrastructure: gateway (1), accessing road, waste control point (2), weighbridge (3), social buildings (4), stripper (5), garbage truck (6), and enclosure (8).
Leachate collection and removal system: drainage system and leachate collection tanks (12).
Landfill gas collection and utilization system: degassing wells, gas utilization station (14), and landfill gas pipeline (7).

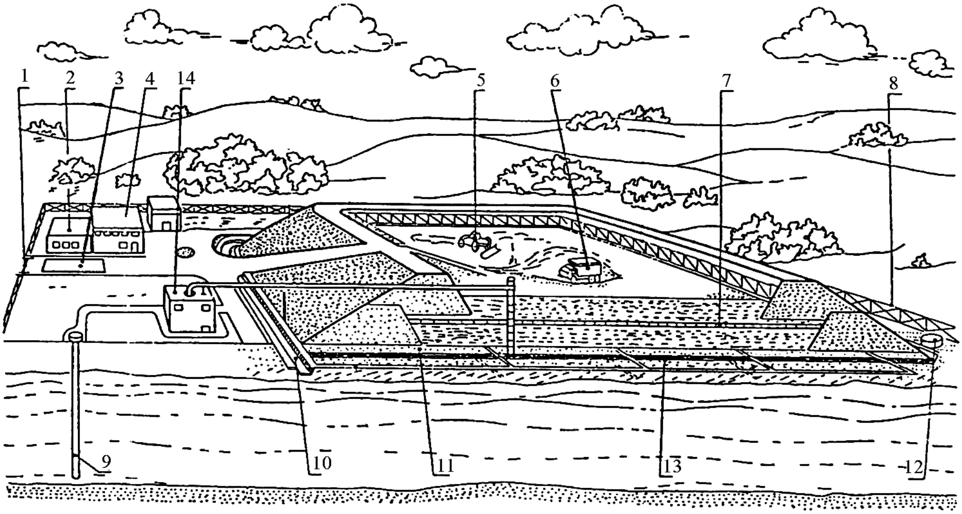

Figure 3. Scheme of the Landfill Site and Surrounding Objects (Zygadło[20]).

Groundwater and surface water monitoring systems: piezometers (9) and surrounding ditch (10).
Base liner system with leachate drainage (11).
Waste layer (13).
Covering system.

Design and construction of landfill, especially its barrier system, should be based on the type of deposited waste and the properties of the ground, where the landfill is located. The hazardous and nonhazardous waste landfills should possess high specification containment barrier liner system to control, contain, collect, and treat the produced leachate and landfill gas. Landfill for inert waste does not require special barrier containment system.

4.2. *Landfill Barriers*

The barrier system insulating landfill in order to minimize its environmental impact is required by the regulations of Waste Landfill Directive.[4] Protection of soil, groundwater, and surface water is to be achieved by the combination of a geological barrier and bottom, side, and cap liner systems during the operational/active phase and the post-closure/passive phase.

The minimum stipulations for the permeability and thickness of the landfill liner system, depending on the category of waste to be deposited in the landfill, are specified. The geological barrier is determined by the geological and hydrogeological conditions below:

- *landfill for hazardous waste*: hydraulic conductivity 1.0×10^{-9} m/s; thickness ≥ 5 m,
- *landfill for nonhazardous waste*: hydraulic conductivity 1.0×10^{-9} m/s; thickness ≥ 1 m,
- *landfill for inert waste*: hydraulic conductivity 1.0×10^{-7} m/s; thickness ≥ 1 m.

Hydraulic conductivity/permeability represents the ease with which a fluid such as leachate will flow through the liner material[1]; this parameter is described by coefficient of permeability (k), Darcy's law $Q = kiA$, where Q is flow rate, i is hydraulic gradient (the pressure difference between the top and bottom of the layers), A is cross-sectional area. Hydraulic conductivities of some natural and synthetic materials are shown in Table 1.

The required characteristics of the geological barrier may be achieved by the use of either natural or synthetic materials. An artificially established geological barrier should be no less than 0.5 m thick.

Table 1. Hydraulic Conductivities of Some Natural and Synthetic Materials (Cited After Williams[1]).

Material type	Hydraulic conductivity (cm/s)
Gravels and gravel-sand mixture	2×10^{-2}
Sands and gravely sands	5×10^{-2}
Silty sands, sand-silt mixture	5×10^{-3}
Inorganic silt and clayey silts	5×10^{-6}
Mixture of inorganic silt and clay	2×10^{-7}
Highly plastic inorganic clay	5×10^{-8}
Compacted clay liner	$1 \times 10^{-6}–10^{-8}$
Bentonite-enhanced soil	5×10^{-8}
Geosyntheic clay liner	$1 \times 10^{-8}–10^{-10}$
Flexibile membranes	1×10^{-11}
Geotextile	$1 \times 10^{-2}–10^{-3}$
Geonet	2×10^{1}
MSW as placed	1×10^{-3}
Shredded MSW	$1 \times 10^{-2}–10^{-4}$
Baled MSW	7×10^{-4}

MSW — municipal solid waste.

Barriers are normally constructed with clay (often bentonite) and a thick plastic membrane. This system is expensive and there is a concern about its long-term performance.

For both hazardous and nonhazardous waste landfill sites, an artificial or synthetic sealing barriers are required, and also the drainage layer, which should be ≥ 0.5 m.

Materials used for sealing layers[1]:

- *Natural clay* — unconsolidated rocks composed of clay minerals that are formed as breakdown products from the weathering of pre-existing rocks. It includes extremely fine grain minerals, such as montmorilonite, illite, and kaolinite, having possibility to absorb the water molecules, what causes swelling. It is used as compacted layer of the 0.6–1.0 m thick.[13]
- *Bentonite-enhanced soil* — mixture of clay minerals, mainly montmorillonite type (bentonite), in the amount of 5–15% and soil (usually sand or silt sands, as they are easier to mix). Bentonite type minerals have a high swelling characteristic on absorbing moisture.
- *Geosynthetic clay liners* — mixture of bentonite clay mechanically or chemically adhered to the geotextile fabric. The bentonite layer may be also sandwiched between two layers of the geotextile fabric, usually 1 cm thick.

Table 2. Properties of Selected Synthetic Flexible Membranes (Cited After Williams[1]).

Property	Butyl rubber	Poli-chlorinated polyethylene	Chloro-sulphonated polyethylene	Ethylene propylene rubber	LDPE and HDPE	PVC
Resistance to UV	good	good	good	good	n.d	poor
Resistance to ozone	good	good	good	good	n.d	poor
Resistance to weathering	good	good	good	good	n.d	poor
Resistance to temperature	good to low and high temperatures	good to low temperatures	good to low temperatures	good to low temperatures	good to low temperatures	poor to low and high temperatures
Resistance to chemicals	low to hydro-carbons	poor to acids and oils	good to acids and oils	poor to oils, hydro-carbons and solvents	good to most chemicals	poor to sulphide
Strength characteristics	low	good	low	high	good	high
Seaming	difficult to seam	easy to seam	difficult to seam	poor seam quality	good seam quality	easy to seam
Other properties	low swelling in water	n.d	n.d	low water absorbance	poor resistance to puncture	n.d

n.d. — no data.

- *Flexible membrane liners* — synthetic and polymeric plastic materials with extremely low permeability. They are characterized by small thickness, high density, favorable mechanical properties, high resistance on UV, chemical and biological conditions, and stress-cracking. The advantages and disadvantages of commonly used flexible membranes are shown in Table 2.
- *Geotextiles* — fabric materials used as protection for polymeric plastic membranes and filtration material to filter out fine-grained particles from the leachate, composed of polypropylene or polyester fibers.
- *Geonets* — porous sheets of plastic netting, used as drainage layers to carry leachate or landfill gas, usually composed of polyethylene.

Residual clay materials from mine overlying layer (waste rock and tailings materials) can be used as the natural barrier or as the synthetic component of the barrier due to the high content of clay minerals. Stabilized sewage sludge and waste composts not complying with the requirements relative to the concentration of heavy metals and microbiological standards for the using in land reclamation can be inserted to the top cover soil in order to improve the growth conditions for plants and microorganisms.

The **base liner system** should protect the environment from the negative impact of landfill leachate percolation and gas emission; however, it should also prevent groundwater seepage into the landfill. The liner system should be resistant to a variety of chemical properties of the leachate. The minimum requirements for the barrier liner system are laid down in the EC Waste Directive[4]; however, due to leakage problems, the design of landfill has resulted in more and more complex liner system.[21] The liner systems rely on the combination of liner materials and leachate and gas collection elements.

There are several types of liner systems[1]:

- *Single liner system* — it consists of only one primary liner (mainly clay layer or bentonite-enhanced soil), which is placed between two separation/protection layers of geotextile (acting as a filter for the fine suspended solids). Leachate collection system would be placed over the upper separation layer; groundwater collection system may lie under the lower separation layer.
- *Composite liner system* — it consists of two different types of liner material, a clay-based material (secondary and lower-laid barrier) and polymeric membrane layer (as primary and upper-laid barrier). The construction is shown in Fig. 4.
- *Double liner system* — it incorporates an intermediate high-permeability drainage layer between the primary and secondary barriers. It is used to monitor and remove

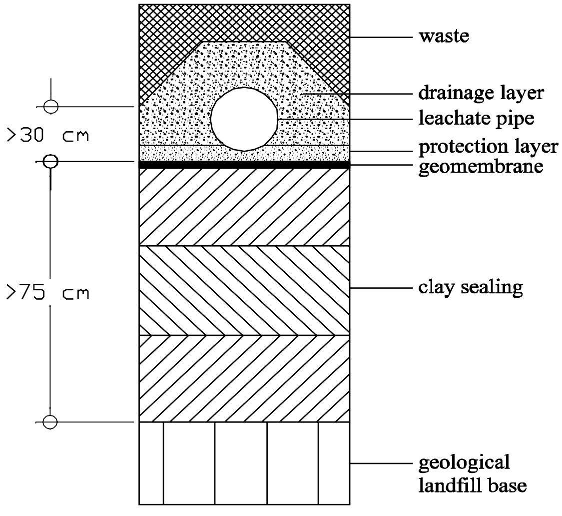

Figure 4. Diagram of a Composite Base Liner System of MSW Landfill (Haubrichs,[22] modified).

leachate and landfill gas from the barriers. Both primary and secondary barriers may be composed of mineral or geosynthetic materials.
- *Multiple liner system* — primary, secondary, and tertiary barriers are incorporated, with intermediate drainage to remove leachate and landfill gas. Primary barrier is usually synthetic polymeric membrane in direct contact with the secondary barrier composed of mineral-based material. The composition of tertiary barrier is not specified.

The risk assessment is the main criterion for the selection of the most appropriate base liner system.

The landfill **covers** can be classified as:

- permanent final covers — they are located on the landfill zone that will not receive any further waste.
- temporal covers (cap) — they are covering areas that will not receive any further waste for a period of at least three months, but for engineering reason, they do not yet have a permanent cap.[23]
- daily cover layers.

Final cover of landfill is a surface barrier between landfill and the environment. According to FRTR Guide for Remediation Technologies Screening Matrix and Reference, the purpose of the landfill caps is to:

- Minimize exposure on the surface of the waste facility.
- Control gas emissions from underlying waste.
- Prevent vertical infiltration of water into wastes that would create contaminated leachate.
- Prevent entry of air, which could disturb the anaerobic biodegradation process.
- Contain waste while treatment is being applied.
- Create a land surface that can support vegetation and/or be used for other purposes.

The design of landfill caps is site specific and depends on the intended functions of the system. Landfill caps can range from a one-layer system of vegetated soil to a complex multi-layer system of soils and geosynthetics. In general, less complex systems are required in dry climates and more complex systems are required in wet climates. An example of cover system design is presented in Fig. 5.

The drainage/gas collection layer consisting of porous material (such as geotextile, geonet, or coarse sand) overlays the waste body. The natural or artificial barrier layer (one or more), lying over the drainage layer, must be protected by geotextile layer above and/or below. Water drainage layer/pipework zone aims at minimization of the amount of percolating water reaching the barrier layer. The drainage protection layer, consisting of soil, can be used to protect the underlying

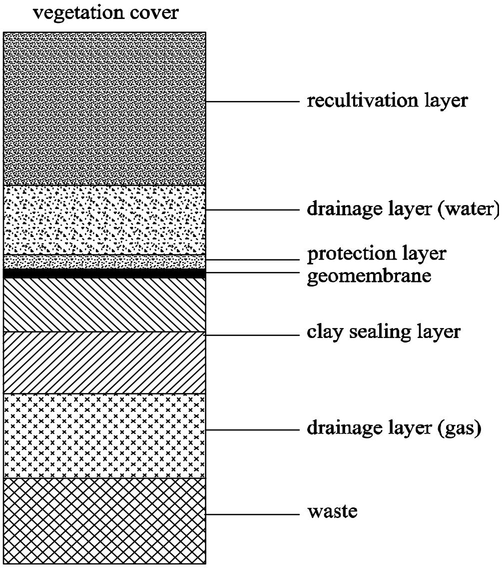

Figure 5. Final Cap Layer of MSW Landfill (Haubrichs,[22] Modified).

system from plant roots. The surface layer/top soil is also called the restoration layer and it should provide proper conditions for plant growth.

Similar to the base liner construction, the materials used for the construction of landfill caps include low-permeability and high-permeability natural materials and low-permeability geosynthetic products. The low-permeability materials divert water and prevent from its passage into the waste. The high-permeability materials carry away water that percolates into the cap. Other materials may be used to increase slope stability.

4.3. *Leachate Collection System and Management*

Leachate represents the water that passed through the waste from precipitation, and water generated from the waste within the landfill site.[1] It is a liquid containing suspended solids, soluble components of the waste, and products from waste degradation processes. The characteristics of the leachate are influenced by the composition of the deposited waste and on the waste decomposition stage.

In addition to the base liner system, a leachate collection system must be added to ensure that the accumulation of the leachate is reduced to a minimum.

The leachate management system consists of:

- leachate drainage,
- collection and pipe-transport system, and
- treatment system.

Leachate collection system is integrated into all liner systems (Figs. 3 and 4). It is composed of sand and gravel or a geonet. In drainage elements of the base and cap barriers, the leachate collection pipes are installed to remove the leachate from landfill body to hold tanks for storage and eventual treatment. In double-liner system, the upper drainage layer is designated for leachate collection, while the lower one is used for leak detection control.

The landfill leachate must be collected and treated to remove pollutants to environmentally acceptable level. The drainage of leachate takes place via gravity flow through drainage gradient paths, which consist of the permeable granular system containing perforated pipes, to collection sumps at low points in the waste mass. Next, the leachate is removed by pumping, gravity drains, or side slope risers at the site perimeter. The leachate treatment is required to remove the components to standards, whereby they can be released to sewer or surface water. Before release, a discharge permit would be required from the local water company or the regulatory authority.

The processes available to treat leachate in order to reduce the pollutants to environmentally acceptable levels may be categorized as physicochemical processes (activated carbon adsorption, reverse osmosis, evaporation, oxidation, wet-air oxidation, air stripping of ammonia, coagulation, flocculation, and settling) and microbiological processes (tricking filters, rotating biological contactors, and aeration in lagoons or tanks). Leachate recirculation back into landfill body and land spreading (for low-contaminant leachate) are also practiced. Throughout the lifetime of the landfill, including post-closure period, sampling and analysis of the landfill leachate are required. This includes monitoring outside the landfill (analysis of groundwater).

4.4. *Gas Collection and Utilization System*

Waste Landfill Directive[4] imposes the requirements for the landfill gas and leachate collection and utilization. A new landfill should be equipped with gas collection well. Landfill gas shall be collected from the landfill receiving the biodegradable waste. The gas must be treated and used. If the gas cannot be used for energy production, it should be flared.

Gas collection system allowing to landfill gas utilization consists of the gas collection wells and connection pipes. Exemplary constructions of the landfill gas collection wells are shown on Figs. 6 and 7.

According to Haubrichs,[22] degasification and landfill gas utilization systems consist of:

- vertical and horizontal collectors (for landfill gas placed in the waste layer),
- gas transport pipes from landfill body to gas stations (substations and central station),

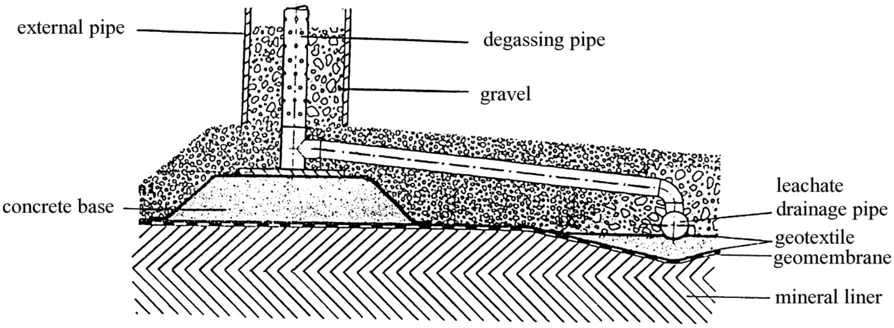

Figure 6. Base of the Gas Collection Well Connected with the Leachate Drainage System (Zygadło[20]).

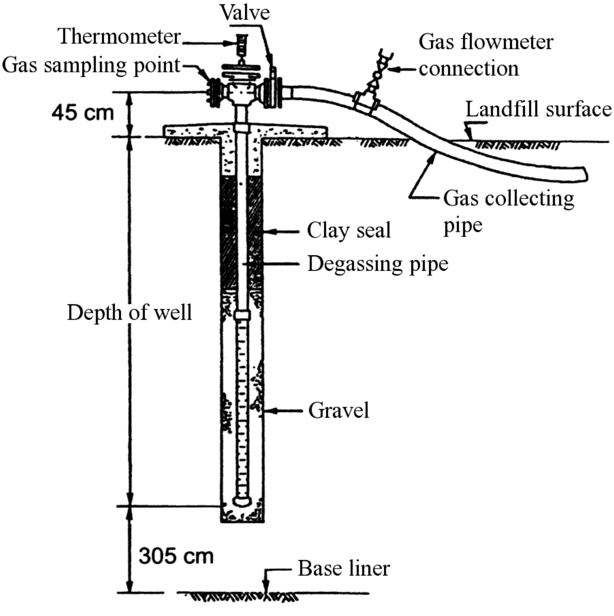

Figure 7. Scheme of the Construction of a Landfill Gas Collection Well (Zygadło[20]).

- condensate trap placed at the deepest point of gas collection system,
- compressor station,
- high-temperature flare and combined heat and power plant, and
- power feeding to grid or own utilization of power.

Landfill gas management can aim at gas recovering for energy production (economic aspect) and simply at reducing the gas emission (environmental aspect). To achieve pollution control and global atmosphere objectives relative to gas production, each landfill site should achieve the highest sustainable position outlined on the hierarchy

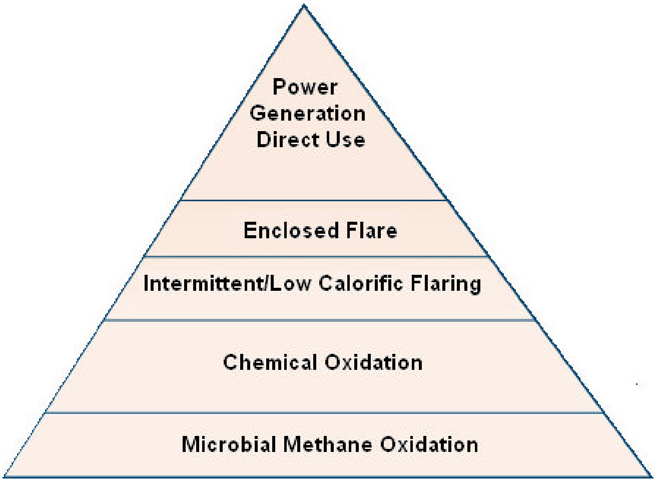

Figure 8. Landfill Gas Management Options (Guidance of Landfill Gas Flaring,[24] Modified).

in Fig. 8 during all phases of a site's operation and post-closure phase. Gas collection with energy recovery is preferred to enclosed flaring. Controlled chemical or biological methane oxidation is at the base of the hierarchy and is appropriate for managing landfill gas in instances of lower gas generation, when a flare cannot be sustained.

Assessment criteria for gas utilization, which should be undertaken at the permit application stage to confirm, whether or not landfill gas utilization is required, are as follows[7]:

- Size of the landfill — sites of less than 200,000 tones of waste are unlikely to produce sufficient landfill gas to generate more than 0.75 MW and are of limited commercial interest.
- Geometry of site — recoveries of landfill gas are highest (up to 75%) on large deep sites. It is often impractical to recover landfill gas from shallow sites or from peripheral areas of the site. It is generally accepted that the minimum depth of site is around 4 m.
- Gas flow rate — between 600 and 750 $m^3 h^{-1}$ — is required to generate 1 MW
- Waste composition — if the waste within the site contains 75% or more inorganic waste, then landfill gas production will be minimal.
- Site location — if a direct use scheme is being considered, then the end user for the gas should be within 10 km of the landfill site; otherwise, the cost of transporting the gas could make the schemes uneconomic.

The landfill gas should be properly prepared before power generation use. The landfill gas clean-up practices presented in the guidance on gas treatment technologies for landfill gas engines[25] include:

(1) establishing practices that already have a role in gas clean-up:
  - after-cooling and prechilling,
  - cyclone separation and other dewatering technologies,
  - particle filtration, and
  - gas engine modifications and other engine management techniques (both in-engine and after combustion) for NO_x, CO, and particulate emission.

(2) emerging and more specialist technologies:
  - wet or dry hydrogen sulfide scrubbing,
  - activated charcoal/zeolites,
  - liquid and/or oil absorption,
  - cryogenic separation,
  - solvent extraction,
  - membrane separation for carbon dioxide, oxygen, and other gas scrubbing/separation techniques,
  - thermal oxidation,
  - catalytic conversion, and
  - in-engine treatment.

There are three categories of compounds, which cause that precombustion gas clean-up is necessary (Table 3).

The raw gas enters the utilization set-up via a dewatering and filtration knockout device that removes moisture and particulates. A gas compressor (or booster) increases the landfill gas pressure to ensure effective operation of the flare or gas engine. Flow metering devices and a slam-shut valve provide the volume flow rate to the flare or engine and act as a final safety control device. The flame arrestors prevent flashback of the flame to the fuel feeder pipe.[25]

Table 3. Contaminants Whose Presence May Require Pre-Combustion Gas Clean-Up.[25]

Category	Reason
Hydrogen sulfide and other sulfur gases	Lead to chemical corrosion of the gas engine (and resultant emission of acidic gases).
Halogenated organics	Lead to chemical corrosion of the gas engine. Potential contribution to emission of acid gases hydrogen chloride (HCl), hydrogen fluoride (HF), and PCDDs/PCDFs.
Silicon compounds	Physical wear caused to the gas engine.

4.5. *Controlled Microbiological Methane Oxidation in Top Covers and Biofilters*

Some of the landfill gas components, such as methane or other hydrocarbons and hydrogen sulfide, passing through the layer of porous material can be oxidized by microorganisms to the less environmentally harmful compounds. The microorganisms responsible for oxidation commonly occur in the natural land and water ecosystems, for example, in the soils, in wetlands, and in paddy rice fields.

Porous material both of the mineral and organic type may be a habitat for the microorganisms using various landfill gas components as an energy or carbon source. The shape of a filtration bed and its volume can be selected depending on the kind of landfill (engineered with the insulated bottom and top cap, and open dumps), predicted volume of the biogas production (of various methane content). Table 4 presents possible technological solutions for landfill gas biofiltration.

Landfill cover soil is a layer put directly on wastes, where diffusion of gases and migration of precipitation water are possible (Fig. 9(a)). The application of this type filtration bed is limited because of the regulations imposing the insulation of landfill cap by the impermeable barrier.[4] According to the Directive,[4] the soil cover, which is the outermost layer of the packing, does not contribute to the reduction

Table 4. Forms of a Biofiltration Bed for Methane Oxidation and their Possible Applications.[26]

Type of a Filtration Bed	Characteristics	Possible Applications
Landfill cover soil	Soil material layer put directly on the waste layer.	Old and inadequately protected landfills of low biogas production or landfills in poor countries. The solution is partial, inexpensive, but easy to apply.
Modified landfill cover	Soil material layer put on an insulated layer of wastes. Biogas moves from the well to the drainage layer and migrates to the oxidizing layer.	Engineered landfills with insulation and a degassing system. Possible application when there is no biogas collection or with the on-ground part of the degassing system has been removed (due to drop of the biogas quality).
Biofilter	Container that is an independent unit for biogas filtration. It can also be installed into the cover as a so-called "bio-window"	Engineered landfills: Small ones — expected short-time production of a high-quality biogas. Big ones — with the biogas collection when their operation for energy production purposes is over, when the decreased quality of biogas makes it useless as a fuel. Big ones — where the amount of organic substances is low — low biogas quality.

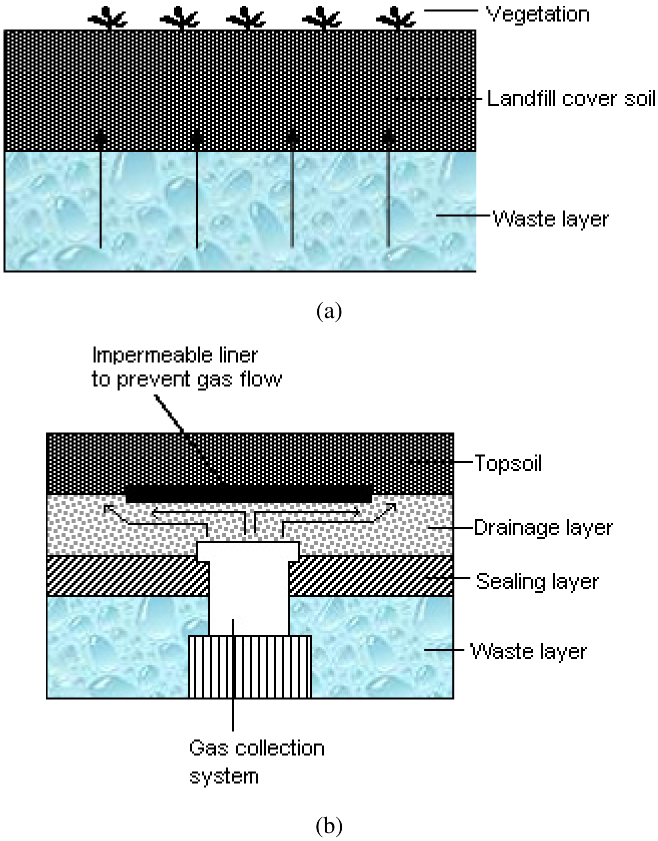

Figure 9. Two Types of a Landfill Cover: (a) Traditional Landfill Cover and (b) Modified Landfill Cover Proposed by Ettala and Väisänen.[27]

of methane and odor emissions but is meant to provide the biological phase of the landfill reclamation possible. Nevertheless, a soil cover can be applied to reclaim old landfills that produce small amounts of biogas. It is a partial but inexpensive solution to the problem of CH_4 and odor emissions in poor countries.

Finnish researchers (Ettala and Väisänen)[27] have proposed an interesting technology that can be applied in the case of landfills equipped with a passive degassing system but where biogas collection is not economically justified. Biogas released from a collection well migrates through the drainage layer to the soil cover and is oxidized therein (Fig. 9(b)).

Many of present day studies on the application of methanotrophs to reduce methane emission from landfills focus on the methane oxidation in biofilters. The biofilter is a chamber of a volume that is determined based on the supplied biogas load size (Fig. 10). The form of a biofilter called "bio-window" can be in-built into

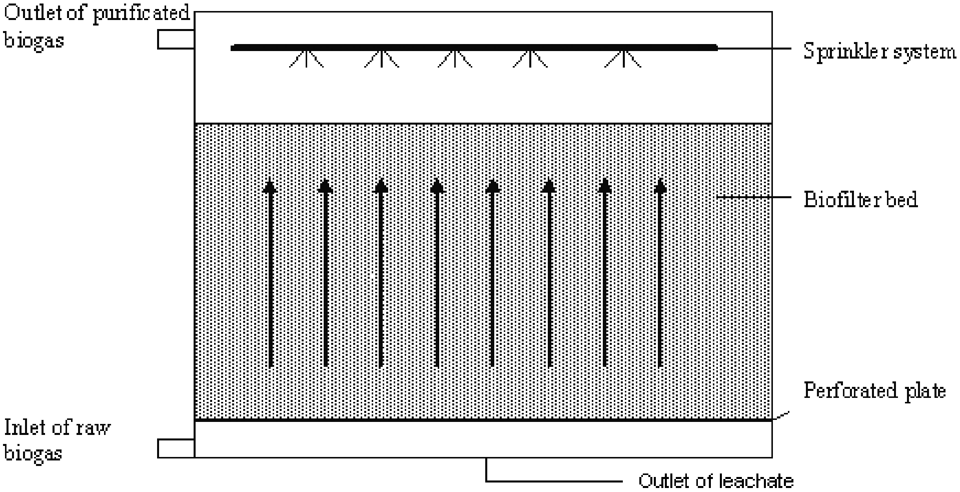

Figure 10. Diagram of a Biofilter for Biogas Purification (Bilitewski *et al.*,[28] Modified).

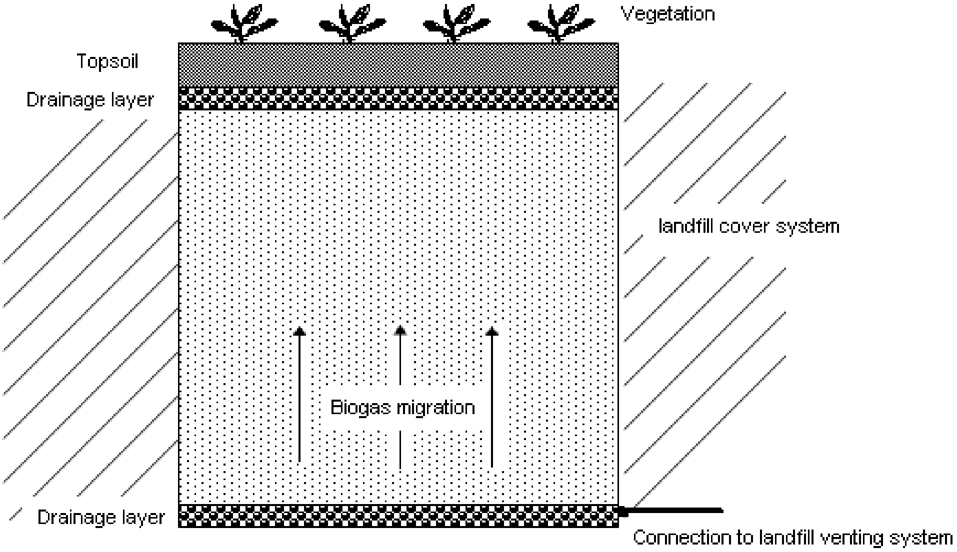

Figure 11. Diagram of a Bio-Window (After Gebert *et al.*[29]).

the cover where part of it has been previously removed (Fig. 11). Biofilter can work with both a passive and an active degassing systems.

The advantage of biofilters operating with active degassing is that it is possible to control the process e.g. by supplying air to the biofilter bottom and by controlling the gas composition. Application of that method is possible in the case of new landfills, the bottom, and the top of which are insulated and where the collection of biogas for energy production is not economically justified. Biofiltration of landfill gas can also be applied as a supplementary method for a biogas recovery system e.g. in a situation when the biogas quality decreases to a value that makes it useless for energy production or when a breakdown of energy generators or a flare occurs.

The application of biofilter bed for the CH_4 oxidation is particularly useful in cases when the biogas emission is low ($<30\,m^3/h$) or when the CH_4 concentration in it is low ($<30\%$). Such biogas is useless as a fuel.[30] As the EU regulations require gradual reduction of the biodegradable fraction deposited in landfills, the installation of degassing systems in municipal dumps will not be profitable in the future. In such a situation, methanotrophy-based solutions seem to be better for municipal landfill because of their low cost and easy application.

5. Daily Waste Landfilling Operation

The following operations may be considered as the elements of daily waste landfilling technology:

- waste weighting,
- scavenging controls,
- waste deposition and planting,
- waste compaction, and
- daily layer capping.

During daily landfill operations, the waste collection vehicles are weighed at a weighbridge on the arrival and their contents is inspected for wastes that do not meet the landfill's waste acceptance criteria. The waste collection vehicles use the existing road network on their way to the tipping face or working front where they unload their load (Fig. 12). Compactors or dozers are used to spread and compact the waste on the working face (Fig. 12). Before leaving the landfill boundaries, the waste collection vehicles pass through the wheel cleaning facility, if necessary, they return to weighbridge in order to be weighed without their load. Through the weighing process, the daily incoming waste tonnage can be calculated and listed in databases. Typically, in the working face, the compacted waste is covered with daily cover. Daily cover is the layer of compressed soil, mineral materials, and inert waste, which is placed on top of a day's deposition of waste. Other form of daily cover is temporary blanket. Blanket can be lifted into place with tracked excavators and then removed the following day prior to waste placement. Chipped wood and chemically "fixed" bio-solids may also be used as an alternate daily cover. The cover prevents from

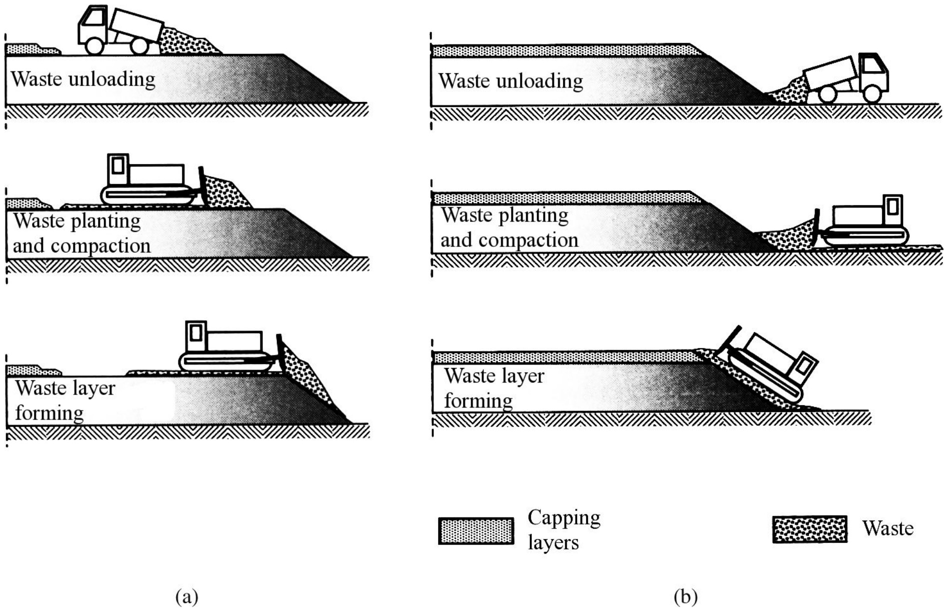

Figure 12. Two Types of the Waste Deposition and Planting: (a) "European" Way and (b) "American" Way.[20]

interaction between the waste and the air, reduces odors and provides a firm base for operation of vehicles.

The space that is occupied daily by the compacted waste and the cover material is called daily cell and it has usually 1–2 m high.

6. Landfill Control and Monitoring Systems

6.1. *Monitoring of Groundwater and Gas Emission*

Waste Landfill Directive[4] obligates to carry on monitoring of gas and leachate composition and groundwater quality throughout the lifetime of the landfill activity and in the post-closure period. Monitoring requirements for waste landfills in the European Union are gathered in the Table 5.

Leachate and gas samples must be collected from representative points throughout the landfill site. Sampling and analyzing the groundwater is required at measuring point in the groundwater inflow area and at two points in the outflow area. Meteorological data (volume of rainfall, ambient temperature, the force of prevailing wind, the evaporation rate, and atmospheric humidity) may also be required,

Table 5. Monitoring Requirements for Waste Landfills in the European Union (Based on Waste Landfill Directive[4]).

Monitoring Task	Operating Phase (Frequency)	Post-Closure Period (Frequency)
Gas emission and leachate data		
Leachate volume	Monthly	Six-monthly
Leachate composition[a]	Quarterly	Six-monthly
Volume and composition of surface water	Quarterly	Six-monthly
Potential gas emission and atmospheric pressure for CH_4, CO_2, H_2, O_2, H_2S, etc.	Monthly	Six-monthly
Groundwater data		
Level of groundwater	Six-monthly	Six-monthly
Groundwater composition	Site-specific frequency	Six-monthly

[a] dependent on the types of waste landfilled.

for example when the potential for build-up of leachate in the landfill site is monitored by the calculation of the balance of water inputs and outputs of the site.

6.2. *Management of Storm Water and Control of Surface Erosion*

Storm water discharge is generated by runoff during rainfall and snowmelt. The discharge often contains pollutants that could negatively impact the water quality of receiving water bodies. The storm water management consists of two elements: drainage control and water detention. The storm water drainage ditches placed around the landfill are designed to effectively convey peak flows and minimize infiltration and sediment transport. Hydrologic studies are used to determine the landfill's watershed design parameters. High density polyethylene and corrugated metal pipelines are used to convey the storm water to concrete channels and desilting basins.

Storm water that should not have the direct contact with landfilled waste is non-contaminated, so there is usually no treatment problem. There are regulations for the quality of the landfill storm water. The discharge limits and monitoring requirements for suspended solids, chemical oxygen demand, and other parameters of the landfill-derived storm water vary in different countries.

The MSW landfill capping constructed of local soils consisting mainly of silt and clay and exposed to severe meteorological conditions such as extreme rainfall events occurring during the rainy season is highly prone to soil erosion. Low hydraulic conductivity of clay and silt topsoil limits infiltration of rainwater and increases generation of runoff leading to intensifying erosion. Sheet, rill, and inter-rill erosions are expected to appear on the landfill top cover. Extensive rainfall events during the rainy season may even create danger of destroying the whole soil layer. In this case, the water drainage layer as well as liner, gas drainage, and even the waste body would be exposed to the atmospheric conditions. Damages caused by water erosion can be also dangerous to gas and water managements of landfill cell. Detailed studies of surface run-off generation and its intensity as well as erodibility of local soils are necessary to predict the water erosion rate and to develop proper soil erosion control and landfill cover soil conservation strategy.

The following necessary countermeasures focused on limiting water erosion of municipal waste landfill cell capping are recommended:

1. Careful selection of landfill capping cross-sectional shape.
2. Protecting the bare soil surface against splash and rill erosion by covering with e.g. straw or bamboo stems.
3. Vegetation development, e.g. grass cover.
4. Implementation of additional erosion control measures such as walls, ribs, and others.

The slope profile of the landfill cell and its geometrical dimensions are directly influencing the velocity of surface water run-off. Short but steep slopes should be avoided in regions of high water erosion risk (soils prone to erosion and extreme rainfall events). The computer modeling of water erosion intensity is a useful tool in the designing process. Many researches showed that the development of vegetation, for example grass cover, highly reduces the erosion rate. Even the use of straw or bamboo stems layer may also slow down the water erosion. Formation of vegetation cover, due to limiting run-off and roots development, strengthens the surface layer of soil, thus slows its water erosion. Additional erosion control measures are useful when natural methods are insufficient. However, in this case, the proper designing calculations are necessary. Nonetheless, careful inspection of capping conditions after every extreme rainfall event and regular observations during the rainy season are necessary.

6.3. *Odor Control*

There are two main sources of landfill odor emission: associated with freshly placed waste and with anaerobic waste decomposition. The main malodorous landfill gas compounds are: organosulfur compounds such as dimethyl sulfide, carbon sulfide, mercaptans and hydrogen sulfide, terpenes (*p*-cymene, *a*-pinene, and limonene), and hydrocarbons.[31] Offensive odor associated with the decomposition of waste is of major concern to landfill operators since it leads to public opposition to landfill site.

The odor management system proposed by Stretch *et al.*[31] integrates various tools in order to support management decision that minimizes odor impacts. It includes:

- An inventory of odor emission sources on the site, including both the quality and quantity of odorants,
- A dispersion model for the analysis and prediction of odor dispersion on and around the site. The model utilizes weather data.
- Models for assessing and predicting the effectiveness of odor mitigation strategies.
- Assessment of performance procedures including a detailed odor complaint logging system and follow-up procedure.

6.4. *Birds, Flies, and Insects Control*

There are several animals that affect safety at landfills. These are birds, rats, flies, and vermin. Gulls, starlings, geese, and pigeons commonly occur around the landfill sites. The studies shown that the populations of gulls can approximately double their number after every two to three nesting seasons.[32]

These are the major health and safety reasons for removal of such birds[32]:

- *technical aspects* — large number of gulls around the face area become close to impossible for the bulldozer operators to even see each other.
- *aircraft safety* — many landfills are located near commercial or recreational airfields in the flight path of such airports. An estimated $5 billion worth of damage and loss of life is caused yearly by birds striking airplanes.
- *health consideration* — the birds might ingest their food on the landfill or take those items off landfill site. This removal of garbage is a large concern as waste material from landfills is transferred to surroundings. If the landfill is close to farming fields, the potential for transplanted disease to animals as well as crop damage is quite large. If the landfill is near residential housing, you could be at risk of spreading several diseases that are caused by the pathogens such as *E. coli* or *Salmonella* spp. Moreover, birds have been identified as carriers of disease, such as the avian flu, which is currently a significant problem in the Far East. When birds are infected, they can live for months, which allow bugs or parasites to transfer disease to other birds, animals, or humans. Landfills provide a place for birds to congregate, feast, and rest.

It is important to adopt bird control plans to protect human health and to prevent damage to airplanes.

The following bird control techniques are in use[33]:

- netting systems,
- pyrotechnics,
- birds of prey,

- distress calls,
- blank firing pistols,
- bird scaring kites,
- automated sound generators,
- rotating mirrors that create multi-colored holograms to confuse the birds, and
- static covering of the working face with soil.

It is difficult to determine which one works best at a particular site.

7. Closure, Capping, and Remediation of Landfill

7.1. *Landfill Closure Procedures*

At the end of the landfill operation, the landfill operator must demonstrate that the site has been physically, chemically, and biologically stabilized and no longer poses a risk to the environment. Stabilization is defined in terms of the quantity and composition of the leachate and landfill biogas. A further consideration is the settlement of the site and the possibility of physical instability of the waste or retaining structures.

Closure procedures will differ depending on whether the entire landfill or only a phase of it is filled to capacity. The following phases of landfill closure are recommended[34]:

- pre-planning: preparing of the final site topographic and landscape plans, site drainage plan, specification of cover material, and vegetation,
- three months before closure: preparing of closure procedures, schedule closing date, and notify appropriate regulatory agencies,
- at closure-erect fences or appropriate structures to limit access, post sign indicating site closure and alternative disposal site, and place cover over the exposed waste, and
- three months after closure — complete drainage and gas control features and structures, install settlement plates, and establish vegetation cover.

The biological phase of landfill remediation begins from the after-use selection and it includes[34]:

- *Determination of cover depth*: Cover soil must be at least 60 cm deep for grass establishment and 90 cm for trees.
- *Establishment of erosion control program*: The soil on recently covered landfills must be stabilized soon after spreading to prevent erosion.
- *Determination of the soil nutrient status*: Before or during the grass and ground cover experiments, soil tests should be made for pH, major nutrients (nitrogen, potassium, and phosphorus), conductivity, bulk density, and organic matter.

- *Determination of soil bulk density*: Cover soil is frequently compacted by landfill equipment during spreading operations to bulk densities that will severely restrict plant root growth.
- *Amendment of soil cover*: The soil that covers the entire planting area should be amended with lime, fertilizer, and/or organic matter according to soil tests before planting. These materials should be incorporated into the top 15 cm of soil.
- *Selection of landfill-tolerant species*: Grasses and other ground covers that will be planted in the soil cover can be selected by evaluating the results of the experimental plots established earlier to determine landfill-tolerant species.
- *Planting of grass and ground covers*: It is generally desirable to embed seed in the soil. Mulches can be used as an alternate to embed the seed but are not so effective.
- *Development of tree and shrub growth*: Trees and shrubs should not be planted until one or two years after grass has been planted. If the planted grass does not grow because of landfill gases, other deeper-rooted species are not likely to thrive either.

7.2. *Remediation of Old Dumps and Landfills*

Since landfilling of waste has occurred throughout history, there are innumerable old landfill sites. According to Jefferis,[35] the problems associated with the old waste landfill sites are as follows:

- Data on waste quantities and composition inputs are usually scarce.
- Old landfills tend to be sited for convenience rather than on grounds fulfilling geological and hydrological criteria.
- Old landfills are often sited close to conurbations, or new developments are close to old landfills.
- The waste degradation process is slow, resulting in long-term generation of leachate and landfill gas after the site is closed.
- Leachate and landfill gas may be migrating from the site.

To protect the environment from old landfill impact special barriers, so-called slurry walls, are created. The creation of the slurry walls is complicated and requires drilling and high-pressure injection of the slurry. The slurry is injected in a series of overlapping "V" cuts, to create an interlocking layer of low permeability barrier. The gas and leachate collection and control systems are introduced by excavating wells or boreholes. The methods of the old landfill lining are very expansive and would only be economically viable in special cases.[1]

New development associated with the remediation of old landfill sites is landfill mining and reclamation. Previously deposited wastes, after stabilization, are excavated for processing to obtain recyclable materials, a combustible fraction and soil as well as a new void space for further landfilling.

7.3. *Possible After Use of Closed Landfill*

Landfill after use is the ultimate use of the land after the landfill permit has been surrendered. It is the one element that should be considered before the start of the landfill restoration process.

According to Interim Guidance of Landfill Closure,[36] the following elements should be also considered:

- landform,
- restoration profile and materials,
- soil suitability (e.g. for methane oxidation, after-use direction),
- detail designs,
- phasing, and
- interim landscape measures.

The main after-use options for the landfills are:

- agricultural use,
- ecological uses,
- recreational and amenity uses, and
- woodland.

In many cases, it is possible — and desirable — for several options to co-exist across the restored landfill.

The factors that influence the choice of after use include:

- Character and quality of available soils,
- Type of waste and associated operational constraints,
- Size, location, and access,
- The development plan or framework,
- The aspirations of local residents, interest group, etc.,
- Scheme economics, and
- Long-term management requirements.

Some of the most common usages of the remediated landfill sites are parks, golf courses and other sports fields, and even office buildings and industrial uses, which are made on a completed landfill. Another strategy for final landfill reclamation is the incineration of landfill trash at high temperature via the plasma-arc gasification process, which is currently used in Japan and will be used at a planned facility in St. Lucie County, Florida.[37]

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Chapter 5

LINERS FOR WASTE

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Abstract

Leachate from waste landfill is attracting attention due to environmental impacts related to pollution of groundwater and surface water. Typically, four groups of leachate pollutants namely dissolved organic matters, inorganic matters, heavy metals, and xenobiotic organic compounds vary in concentration during acid phase and methanogenic phase required landfill to be lined. Several types of liners consist of clay, geomembranes, geotextiles, geosynthetic clay liner, and geonet are generally used in municipal solid waste (MSW), which have to take into considerations of hydraulic conductivity, shear strength, chemical compatibility, and resistant to creep and puncture.

Keywords: Leachate, landfill, pollutants, dissolved organic matters, inorganic matters, heavy metals, xenobiotic organic compounds, clay, geomembranes, geotextiles, geosynthetic clay liner, geonet, environmental pollution, liners for waste.

1. Landfills

1.1. *Type of Landfills*

Landfill is generally dumpsite used for the disposal of solid wastes. It is the cheapest and most simple method for the disposal of wastes subject that adequate land is made available for the intended use of the landfill. However, a dumpsite used for the disposal of wastes by the landfilling method has to be carefully selected and designed for its intended use. Normally, wastes that are disposed in the landfill have to be compacted to reduce volume and to be covered at the end of day. Of concern with the use of landfill may result in various environmental issues such as leachate generation. In general, there are three types of landfill used for waste management purposes. The types of landfill are categorized as depicted in Table 1 according to waste types.

1.2. *Leachate Generation*

Leachate is the liquid percolation that drains through the waste in the landfill varies widely depend on waste type and the waste age. Typically, the leachate can be characterized into four major groups as shown in Table 2. The four major groups are mainly dissolved organic matters, inorganic matters, heavy metals, and xenobiotic organic compounds. Beside these, other compounds, such as arsenate, barium, borate, cobalt, lithium, mercury, selenate and sulfide, are also likely present in the leachate, however, in small quantity and of less significant level.

Leachate can be generated by several potential sources include gravity drainage, ponded water, rain, infiltration, and groundwater inflow. Leachate percolating waste above groundwater table causes contaminant migration to groundwater from waste whereas leachate leaches through waste at near shore and past contaminant transport known as tidal flushing.

The dissolved organic matters are organic molecules of varied origin and composition in leachate that are measured in terms of biochemical oxygen demand (BOD),

Table 1. Type of Landfill.

Type of Landfill	Description
Type I Inert landfill	Landfill is used for waste that is not interacted with other substances
Type II Noninert and nonhazardous landfills	Landfill is lined and is equipped with leachate collection system, air pollution system, and is compartmented. It is typically meant for domestic waste and nondangerous industrial wastes
Type III Hazardous landfill	Landfill is used for dangerous and hazardous waste

Table 2. Pollutants in Leachate.

	Group of Pollutants in Leachate	Components
1.	Dissolved organic matters	Acids, alcohols, aldehydes, and others usually quantified as COD, TOC, other volatile fatty acid, and refractory compound include fulvic- and humic-like compounds
2.	Inorganic matters	Sulfate, chloride, ammonium, calcium, magnesium, sodium, potassium, hydrogen carbonate, iron, and manganese
3.	Heavy metals	Lead, nickel, copper, cadmium, chromium, and zinc
4.	Xenobiotic organic compounds	Aromatic hydrocarbon, phenols, chlorinated aliphatics, pesticides, and plastizers include PCB, Dioxin, PAH, etc.

total organic carbon (TOC) and chemical oxygen demand (COD). Generally, BOD/COD ratio is used to describe the organic composition in the leachate.

The inorganic matters such as ammonia, calcium, chloride, hydrogen carbonate, iron, magnesium, manganese, potassium, and sulfate are found in most landfill leachate and mostly experience wash out in landfill instead of sorption and precipitation. Heavy metals such as cadmium, chromium, copper, lead, nickel, and zinc are found generally low; however, varied from different landfills. Xenobiotic organic compounds such as aromatic hydrocarbon are found to be particularly low in concentration in municipal landfill leachate.

In the landfills, wastes once contained decompose at least in four phases comprising various biological and chemical reactions as depicted in Table 3. The four phases are the aerobic phase, the anaerobic acid phase, the early methanogenic phase, and the mature anaerobic acid phase.

In the early days of landfill usually known as the aerobic phase, the oxygen present in the waste is consumed rapidly resulting in the increase of waste temperature and carbon dioxide. Leachate is also generated due to precipitation of the waste in the landfill cell.

Subsequent to the aerobic phase, fermentation reaction takes place, as waste in the landfill cell becomes anaerobic. In this anaerobic phase, the biodegradable contents in the waste are decomposed and are converted to methane and carbon dioxide with the aid of bacteria. These bacteria are mainly the fermentative bacteria that ferment and convert monosaccharides to alcohols and carboxylic acids. The acetogenic bacteria present subsequent convert these alcohols and carboxylic acids to acetate, carbon dioxide, and hydrogen and the methanogens then convert these products to carbon dioxide and methane.

In the presence of methane, the waste becomes neutral and the acids accumulated are converted to carbon dioxide and methane by methanogenic bacteria in the early methanogenic phase. The methane generation rate will increase to maximum

Table 3. Phases of Waste Decomposition in Landfill.

Waste Decomposition Phases	Waste Decomposition Reaction
1. Aerobic phase	• Occur only in early few days as long as oxygen is available • Leachate mainly generated from waste compaction and precipitation
2. Anaerobic phase	• Occur only once when oxygen in the waste is depleted • Leachate generated after fermentation reaction on waste by bacteria to methane and carbon dioxide under anaerobic conditions • Highest COD and BOD are anticipated and are mainly acidic
3. Early methanogenic phase	• Occur when sufficient quantities of methane is generated and pH is approaching neutral due to conversion of acid to methane and carbon dioxide by methanogenic bacteria • COD and BOD values begin to reduce and COD to BOD ratio increases, as soluble substrate is depleting
4. Mature methanogenic phase	• Occur as methane generation reaches its highest rate, as soluble substrate is significantly reduced • pH continues to increase and COD to BOD ratio increases tremendously due to the highest consumption of soluble substrate

level and decrease subsequently until the depletion of soluble substrate in the stable methanogenic phase. In this phase, the BOD/COD is anticipated to reach 0.1 or lower, as soluble substrate is consumed and exhausted.[1]

Once the landfill is full and final cover is placed, the decomposition of waste is still on going and the generation of leachate is anticipated to decrease as time go. It is usually presumed that the landfill will be stable after 30 years from the closure.

Typically, the composition of leachate generated from the landfill is subject to waste age and other factors such as waste type and landfill approach used. As leachate percolates through waste strata layers that undergo various decompositions, several studies had reported that leachate may contain high amount of both dissolved organic matters and inorganic matters with average concentration of thousand folds higher than those found in groundwater.[2,3] The concentration of these pollutants may vary over phases of waste decomposition in landfill as leachate generated is anticipated to have low pH and high concentration of ready biodegradable organic pollutant in the early acid phase followed by early methanogenic phase with high pH and lower concentration of biodegradable organic content to the later mature methanogenic phase with higher pH and in waste. This is represented by higher BOD/COD ratio in the acid phase as compare to the methanogenic phase.

It is also reported that water content in the landfill will affect the rate of waste decomposition and the time taken for methane generation to reach zero.[4] Waste decomposition is anticipated to be slower in dry weather condition with infiltration

Table 4. Composition of Landfill Leachate.[5]

Parameter[a]	Range
Heavy metals	
Arsenic	0.01–1
Cadmium	0.0001–0.4
Chromium	0.02–1.5
Cobalt	0.005–1.5
Copper	0.005–10
Lead	0.001–5
Mercury	0.00005–0.16
Nickel	0.015–13
Zinc	0.03–1000
Inorganic macrocomponents	
Total phosphorous	0.1–23
Chloride	150–4500
Sulfate	8–7750
Hydrogenbicarbonate	610–7320
Sodium	70–7700
Potassium	50–3700
Ammonium-N	50–2200
Calcium	10–7200
Magnesium	30–15000
Iron	3–5500
Manganese	0.03–1400
Silica	4–70
pH	4.5–9
Spec. Cond. (μS cm^{-1})	2500–35000
Total solids	2000–60000
Organic matter	
TOC	30–29000
BOD_5	20–57000
COD	140–152000
BOD_5/COD (ratio)	0.02–0.80
Organic nitrogen	14–2500

[a] Values in mg/l unless otherwise stated.

of 500 m or less. In most engineering design, recycling of leachate in landfill is used to boost the water content up to 50% and to ensure the presence of sufficient quantity of substrate and bacteria.

The composition of typical leachate quality is depicted in Tables 4 and 5,[5] which illustrate changes of several parameters over time as the waste is decomposed.

Dissolved organic matter in leachate consists of various organic degradable constituents ranging from small volatile acids to refractory fulvic- and humic-like

Table 5. Composition of Leachate During Acidic Phase and Methanogenic Phases.[5]

	Acidic Phase		Methanogenic Phase		
Parameter	Average	Range	Average	Range	Average
pH	6.1	4.5–7.5	8	7.5–9	
BOD_5/COD (ratio)	0.58		0.06		
BOD_5	13000	4000–40000	180	20–550	
COD	22000	6000–60000	3000	500–4500	
Sulfate	500	70–1750	80	10–420	
Calcium	1200	10–2500	60	20–600	
Magnesium	470	50–1150	180	40–350	
Iron	780	20–2100	15	3–280	
Manganese	25	0.3–65	0.7	0.03–45	
Ammonium-N					740
Chloride					2120
Potassium					1085
Sodium					1340
Total phosphorus					6
Cadmium					0.005
Chromium					0.28
Cobalt					0.05
Copper					0.065
Lead					0.09
Nickel					0.17
Zinc	5	0.1–120	0.6	0.03–4	

compound.[6] Higher dissolved organic matter is anticipated in the acid phase when compared to those in methanogenic phase. Like the dissolved organic matter, the concentration of inorganic matter is much depended on the stabilization of landfill. It is also reported[7] that the inorganic matters are lower in methanogenic phase due to lower dissolved organic matter and higher pH.

As waste in the landfill contains organic matter that has good sorptive capacity for metal immobilization, the presence of heavy metals in the leachate is anticipated to be relatively low. The low concentration of heavy metal is probably due to the presence of sulfide formed from sulfate reduction during waste decomposition.[8]

Wide spectrum of xenobiotic organic compounds are found in landfill leachate depends on waste composition, waste age, and landfill approach. Typical xenobiotic organic compounds are halogenated hydrocarbons and monoaromatic hydrocarbons.

In general, leachate composition is very much depend on waste composition, waste age, and if the landfill is lined. Leachate of acid phase as compared to methanogenic leachate is anticipated to have higher concentration of both organic and inorganic pollutants that leach through the underlying strata.

1.3. *Environmental Impacts*

Generally, environmental issues related to improper landfills are groundwater pollution and soil contamination. Once waste is buried is a landfill, the action of ever-present water cause many physical, chemical, and biochemical reactions. Leachate is produced when a sizeable portion of the buried wastes in the landfill becomes saturated with water from external sources. When waste decomposes with the action of water, the resulting leachate percolates downwards. As it does so, it absorbs more chemical compound and microorganisms naturally present in the waste. The constituents of leachate can be categorized into four types; namely, (i) dissolved organics matter such as volatile organic compounds and total petroleum hydrocarbons, (ii) inorganic matter such as acids, sulfides, and chlorides, (iii) heavy metals such as Cu, Fe, Sn, Pb, Cd, B, and Hg, and (iv) xenobiotic organic compounds such as aromatic hydrocarbon and dioxins. The chemicals and microorganisms contained in leachate are potentially harmful due to gradual degradation of subsurface water. They can cause adverse impacts to the environment and endanger life. Even under controlled condition, such as a well-planned and well-run sanitary landfill, leachate may percolate or penetrate through the natural ground and contaminate groundwater and underground freshwater supplies. The environmental impact is significant particularly those without engineering control such as liners and leachate collection system.

Naturally, the ground has a way of neutralizing infiltration of chemicals and organisms contained in leachate. This is done by weakening the content or the amount of contamination percolate by the leachate as it drains down the soil. The major potential effects of these environmental impacts include the depletion of oxygen to natural water sources and toxicity caused to fauna and flora in the aquatic environment. Six occurrences are anticipated to moderate leachate.

First, the infiltration of water in the soil dilutes leachate. Second, the soil may absorb the pollutant, so that they stick in extremely thin films of molecules to the surfaces of solid or liquid particles in the soil. Third, temperature and naturally occurring chemicals in the soil may cause the pollutant to precipitate or separate from the leachate solution. Forth, the soil serves as filter of suspended particles from the leachate. Fifth, ion exchanges occur due to electrical charges of soil particles, resulting in the removal or separation of minerals and other substances dissolved in the leachate. Finally, leachate is diluted by dispersion being spread out over a wide surface area.

Various factors may also react with leachate during percolation that may yield changes in chemistry and pollutant strength reduction as shown in Fig. 1. These factors maintain the forms of physical forces such as filtration, sorption, advection,

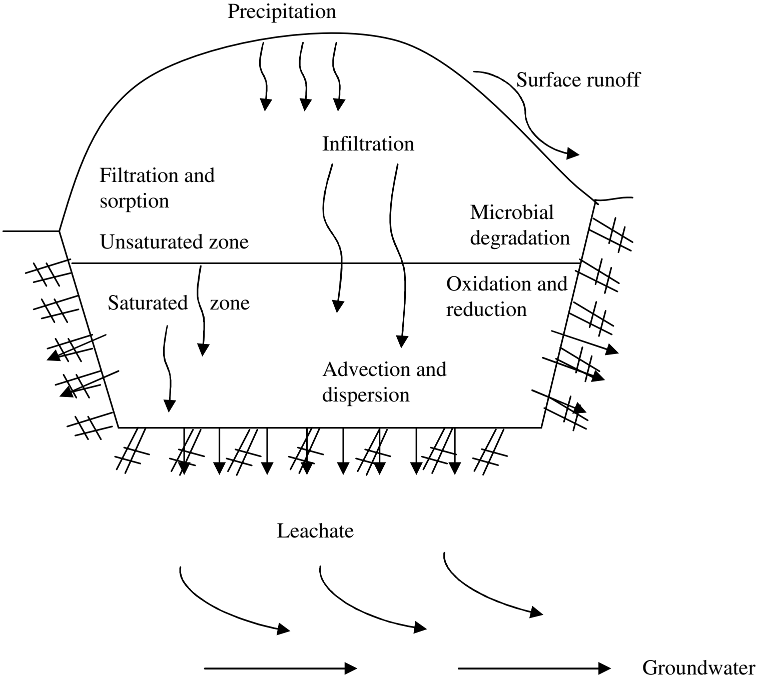

Figure 1. Leachate Migration in Landfill.

and dispersion, chemical forces such as oxidation–reduction, precipitation–dissolution, adsorption–desorption, hydrolysis, and ion exchange, and biological forces such as microbial degradation. However, these reactions are affected by ground material, ground hydraulic condition, and leachate chemistry. Even though these reactions have the capacity to reduce pollution impact, it is also anticipated that some reactions such as microbial degradation can cause negative impact such as the increase of toxicity from the original pollutant.

2. Liners

Liners are commonly used in applications ranging from landfill covers and bottom liners to secondary containment systems, decorative ponds, and wastewater lagoon. In landfills, liner used to minimize contamination from the landfill by controlling and isolating the leachate generated in the landfill as shown in Fig. 2. Various types of liners can be used for landfill sites. However, several considerations shall be given in the selection of the liners for the landfill site include permeability, strength factor, and shear factor.

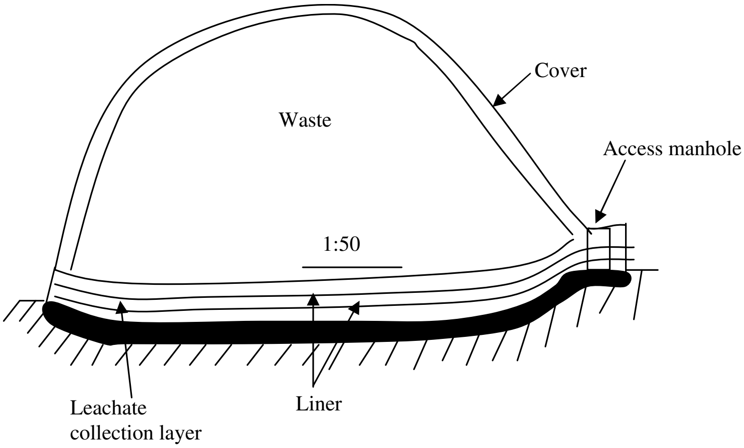

Figure 2. Typical Landfill with Liner.

2.1. *Types of Liners*

Landfill liners are served as barrier between the waste in the cell and the surrounding environment and to channel off the leachate to collection and treatment facilities; therefore, they prevent water pollution. Thus, the main function of liner is to provide an impermeable barrier using various materials with high values of elastic modulus, chemical and weathering resistances, yield, and puncture strength. The materials used include clay, geomembranes such as HDPEC (high-density polyethylene), PVC (polyvinylchloride), PP (polypropylene), geotextiles, geosynthetic clay liner, and geonet as shown in Table 6.

Clay is one of the most economical liner materials used in most landfill application due to its low permeability, low diffusivity, ductility, chemical compatibility, chemical retardation, internal and interface shear strengths, and good constructability. However, it is affected by the factors such as construction requirement, soil composition, and post-construction changes.[9]

Modified clay such as bentonite is used to replace clay due to its mineralogy usually in term of percentage of sodium and/or calcium montmorillonite, moisture content, and operation requirement to improve permeability.

Synthetic liners are used as an alternative to clay such as geomembrane and geotextile because of low volume consumption and easy availability. The rapid acceptance of synthetic liners in the engineering application is due to their high strength, chemical compatibility, and thicknesses up to as thin as 1 mm.

For selection of an effective liner for landfill, important criteria include hydraulic conductivity, shear strength, chemical resistance, and other performance characteristics such as free and confined swelling and rate of creep. Liner with low hydraulic

Table 6. Components of Liners.

Component	Materials	Advantages	Disadvantages
Clay	Compacted clay	Good for groundwater protection from clay	Fracture can be caused by chemical attack, drying out, and freezing-thawing
Geomembranes/ flexible membrane liner (FML)	Made of various plastic materials including PE, PVC, PP, and HDPE	Strong, high chemical resistant, and impermeable to water	Clogging due to trap particles
Geotextiles	Made of woven or nonwoven textile sheeting	Effective water movement	Clogging due to trap particles
Geosynthetic clay liner (GCL)	Made of thin clay layer between two layers of geotextiles	Easy installation	Less impact by freezing-thawing
Geonet	Made of plastic net	Effective water movement	Clogging due to trap particles

conductivity is to ensure low permeability (i.e. rate of infiltration) through waste and strong shear strength is to ensure maximum stress of liner without losing structural integrity.

Geosynthetic clay liners (GCL) are one of the newest liner technologies used in municipal solid waste (MSW) landfill application due to their low hydraulic permeability, easy installation, and swelling property. Typically, the configuration of GCL consists mainly of modified clay i.e. bentonite either sandwiched between two sheets of geotextile or bonded to a geomembrane as shown in Fig. 3. A geotextile that is woven or nonwoven sheet material is less impervious to liquid and more resistance

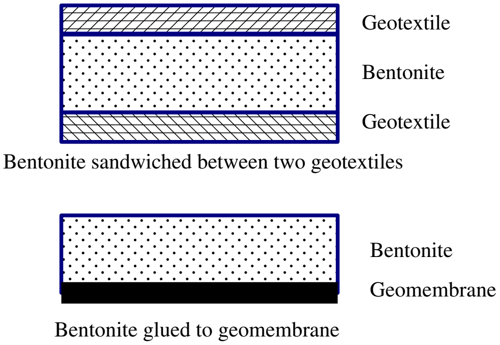

Figure 3. Typical Configuration of GCL.

Table 7. Comparison of CCL and GCL.(10)

Compacted clay liner	Geosynthetic clay liner
Thick (0.6–1.5 m)	Thin (<10 mm)
Field constructed	Manufactured
Hard to build correctly	Easy to build (unroll and place)
Impossible to puncture	Possible to damage and puncture
Constructed with heavy equipment	Light construction equipment required
Often required test pad at each site	Repeated field testing not needed
Site-specific data on soil needed	Manufactured product: data available
Large leachate-attenuation capacity	Small leachate-attenuation capacity
Large thickness, takes up space	Little space is wasted
Cost is highly variable	More predictable cost
Soil has low tensile strength	Higher tensile strength
Can desiccate and crack	Cannot crack until wetted
Difficult to repair	Not difficult to repair
Vulnerable to freeze-thaw damage	Less susceptible to freeze-thaw damage
Performance depends highly on quality of construction	Hydraulic properties are less sensitive to construction variabilities
Slow construction	Much faster construction

to penetration damage when compared to a geomembrane that is a polymeric sheet material.

GCL offers a good substitute to other conventional landfill liners such as CCL (compacted clay liner) due to fast and easy installation, low permeability, low conductivity, good swelling properties, and cost effective in the absence of clay material as depicted in Table 7.[10] Modified clay or bentonite of GCL is an excellent absorbent that attracts positively changed water particles and hydrates rapidly when exposed to leachate that maximizing capacity at the same time better environment protection.

Typically, there are five types of architectures used for landfill liners that can be described as single, double, and composite as shown in Table 8.

A *single liner* normally consists of a clay liner or CCL, a geomembrane or a GCL is used in landfill for construction, which is cost-effective to build and maintain.

A *composite liner* consists of the combination of geomembrane with clay liners so as to limit leachate migration, which is usually used in MSW landfills.

A *double liner* consists of either two single liners, two composite liners, or a combination of a single and a composite liners in such a way that the upper liner can collect the leachate and the lower liner can act as back up for leakage. Double liners are used in either MSW landfills or hazardous waste landfills.

In addition, a leachate collection system consists of sand and gravel is used to drain the leachate from the landfill to collection ponds for storage and treatment. In addition, a protective layer consisting of soil, sand, and gravel or a layer of soft

Table 8. Types of Liners.

Liner Type	Composition	Function	Application
Single liner	Clay liner, geosynthetic clay liners, or Geomembranes	Sufficient to prevent insoluble leachate migration	Suitable for domestic solid waste and nondangerous industrial waste landfills
Single composite liner	Two or more different materials of low permeability such as clay liner with geomembrane	Effective to control leachate migration with clay liner or geomembrane	Suitable for MSW and nondangerous industrial waste landfill
Double liners	Two single liners, with two composite liners or a single liner with a composite liner	Primary liner is to collect leachate while secondary liner serves as back up	Suitable for hazardous waste landfill
Double composite liner	Two composite liners place one above the other	Ensure sufficient collection of leachate and no leakage	Suitable for hazardous waste landfill

waste (e.g. organic waste, paper, rubber, and others) is used to cushion liner to avoid damage as shown in Fig. 4.

2.2. *Liner Failure*

Generally, the two failure modes of liners are leakage and liner destruction. Leakage occurs in liners through lost of material permeability or hole damage cause leachate or even waste to release to the environment. Liner destruction on the other takes place due to extensive membrane movement or loss of mechanical properties caused by phenomena including creep and puncture.

Creep is defined as a deformation of material over a prolonged period under constant pressure.[11–12] It is load, temperature, and time dependence and is related to most mechanical deformation such as compression, tensile, torsion, and flexure. However, only compressive and tensile creeps are anticipated in the geomembrane due to material used. The three phases of creep behavior observed are: (i) primary with strain increases but strain rate decrease, (ii) secondary with both strain and strain rate remain constant, and (iii) tertiary with material rupture due to rapid increase of strain and strain rate. The typical creep curve for landfill liner is depicted in Fig. 5.

Puncture is one of the most common and serious types of damage to the landfill liner. Puncture phenomena cannot be assessed easily and are subject to short-term as well as long-term puncture forces. Short-term forces normally occur during installation of leachate collection layer while long-term forces occur due to over burden loads of waste.

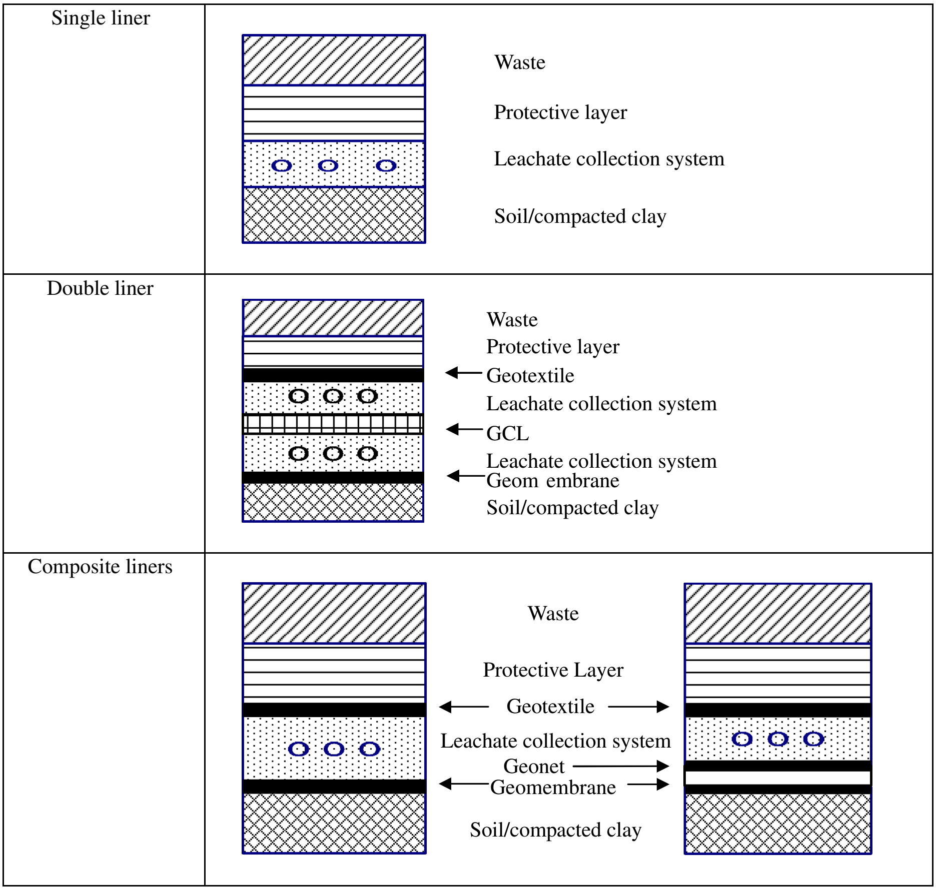

Figure 4. Cross Section of Liner.

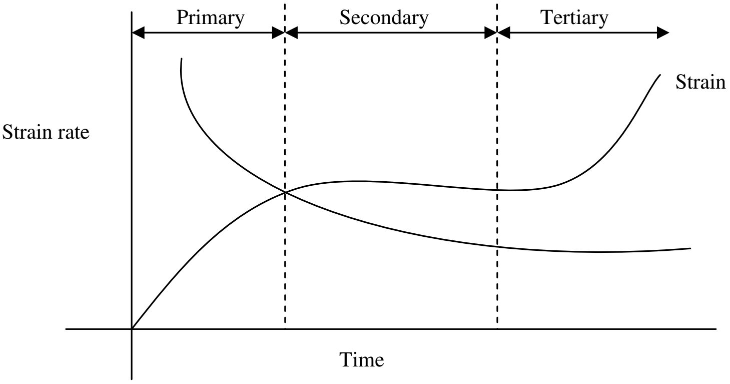

Figure 5. Typical Creep Curve.[12]

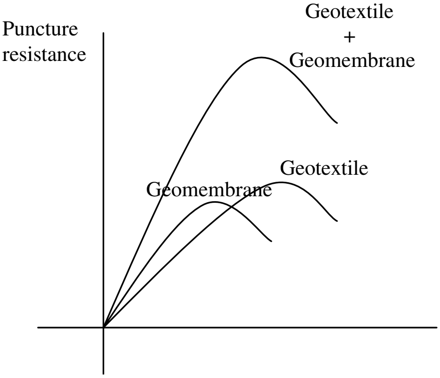

Figure 6. Puncture Resistance of Liners.[13]

There are two types of puncture phenomena, namely static and dynamic. The liner usually experiences dynamic puncture phenomenon due to fall height during installation and is usually short term. Static puncture phenomenon is due to contact with static normal stress and can be short term such as traffic load and long term such as waste load. Figure 6 depicts the puncture resistance summation of two different components of liner.[13]

2.3. *Liner Design*

Typically, liner design involves the material selection and thickness computation based on various factors including type of pollutants, environmental and climatic conditions, material availability, legal requirements, and cost provision. The composition of liners for waste usually consists of a compacted subsoil layer overlain by compacted clay layers and geosynthetic materials with different combination of materials all depend on application requirement.

The design flow of liner as depicted in Fig. 7 is broadly covered in five steps. Each of the design steps must be documented in detail so that decision can be made to finalize design for application.

Step 1 *Data Collection and Evaluation* — The collection of data pertaining to waste characterization such as waste type and strength and site geometry such as water bodies and chemistry must be conducted for engineering evaluation to establish benchmarking references for liner system and technique selection.

Step 2 *Selection of Liner System and Construction Technique* — Relevant screening on the type of liner material for leachate control and cover design to minimize pollutant release that meet acceptable limit must be carried out. In addition, the study of various available construction technologies for liner placement must be

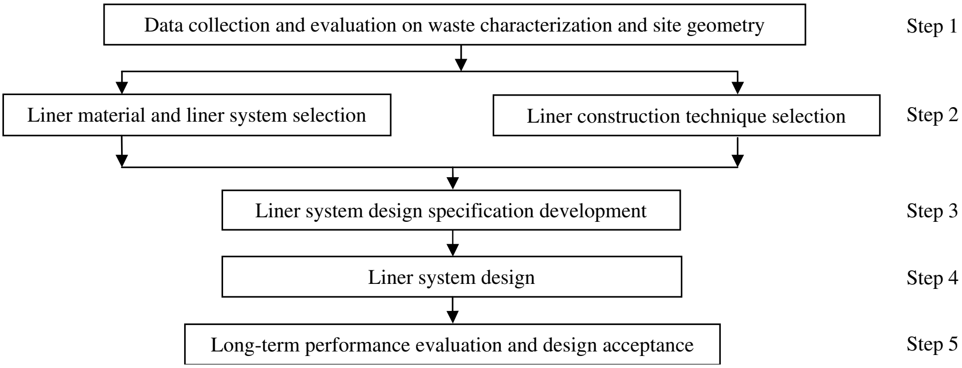

Figure 7. Design Flow for Liner System.

conducted to ensure accurate placement of liner, so that necessary density and rate of liner application can be derived.

Step 3 *Liner System Specification Development* — Next is to develop design specification, so that the design effectiveness of the liner system is ensured. The design specification is to be outlined, so that the liner system constructed can meet specific performance criteria at minimum cost.

Step 4 *Liner System Design* — The liner system is then designed with appropriate engineering criteria for reducing pollutant release thus achieve the intended liner function. The design involves mainly the determination of liner thickness required taking into consideration of physical and chemical properties of pollutant and the material of liner selected.

Step 5 *Long-Term Performance Evaluation and Design Acceptance* — It is crucial for the liner system designed to be evaluated for long-term performance on its inherent ability to maintain the design effectiveness throughout the entire life cycle of the liner system. The design of liner system is accepted for application as long as the compatibility of liner material and pollutant present is reached, so that design characteristic can be assured for extended period of time.

Table 9 illustrates the template developed for specific guidance for design of liner system for waste landfill.

Adequate considerations are to be given for liner system design to prevent leakage, geogrids for leak collection, and a good protection layer to prevent puncture phenomenon and to reduce creep, stress cracking, and aging phenomenon. The design consists of site selection, geometric layout, geotechnical consideration, cross-sectional determination, geomembrane material selection, thickness determination, side-slope and cover soil details, anchor trench details, seam type decision, seam testing strategy, design of connection and appurtenances, leak scenarios and correction measures, and proper quality assurances.[14]

Table 9. Liner System Design Template.

1. Baseline data	• Waste source • Waste characterization — pH, conductivity, total solids — Organic matters — Inorganic matters — Heavy metal • Site geometry and environment — Geotechnical aspect — Groundwater and surface water — Climate • Legal requirement — Legislative and regulatory
2. Liner system selection	• Liner materials resistant — Compacted modified subsoil — Compacted clay — Geomembranes — Geosynthetic — Composite materials • Liner system — Leachate collection system — Cover — Wall and trenches • Engineering techniques — Compaction and recompaction — Lift — Water content
3. Liner system design specification	• Liner thickness • Liquid and flow through • Hydraulic conductivity • Strength and puncture resistance • Shrink-swell properties
4. Liner system design	• Liner thickness • Pollutant flux • Pollutant removal
5. Long-term performance evaluation	• Design effectiveness • Liner compatibility

Problems selecting to the liner design are identified to guide designing engineer as shown in Table 10.[15]

Design features are to take into consideration of a safety analysis for liner system as depicted in Table 11.[16]

Table 10. Problem Associated With Liner Design.[15]

Problem	Liner Stress	Required Properties: Geomembrane	Required Properties: Landfill	Typical Factor of Safety
Liner self-weight	Tensile	G, t, σ_{allow}, δ_{L}	β, H	10–100
Weight of filling	Tensile	t, σ_{allow}, δ_{L}, δ_{u}	β, h, H, γ	0.5–10
Impact during construction	Impact	I	D, W	0.1–5
Weight of landfill	Compression	σ_{allow}	γ, H	10–50
Puncture	Puncture	σ_{p}	γ, H, P, A_{p}	0.5–10
Anchorage	Tensile	t, σ_{allow}, δ_{L}, δ_{u}	β, γ, Φ	0.7–5
Settlement of landfill	Shear	τ, δ_{u}	β, γ, H	10–100
Subsidence under landfill	Tensile	t, σ_{allow}, δ_{L}, δ_u, χ	α, γ, H	0.3–10

Note: Geomembrane properties: G = specific gravity, T = thickness, σ_{allow} = allowable strength, T = shear strength, I = impact resistance, σ_{p} = puncture strength, and δ_u = friction with material above. Landfill properties: β = slope angle, H = landfill height, γ = unit weight, h = lift height, α = subsidence angle, Φ = friction angle, and d = drop height.

2.4. *Liner Installation and Maintenance*

The criterion to be selected in engineering practice for installation and maintenance is crucial to ensure effective use of liner in landfill.

Proper installation of landfill liner is to be carried out with care by competent workmanship to follow the design work. A proper and detail quality control and management plan are to be put in place for operation throughout the entire lifespan of the landfill liner system to monitor long-term performance with respect to liner integrity and landfill stability.

2.5. *Prevention of Contaminant Migration*

The latest engineering design of landfill system is to limit contaminant migration so as to minimize impact to surrounding environment. The typical landfill system consists of a leachate collection system to control the leachate head acting on the liner and to collect and remove leachate. The leachate collection system comprises mainly a genotextile with granular layer and perforated pipes. The landfill liner can range from a clay layer to a liner system of one or more geomembrane and/or compacted clay liner or geosynthetic clay liner. Composite liners comprising of a geomembrane over compacted clay have been widely adopted in many standard landfill designs [9]. The effectiveness and efficiency of liner performance are based on the several factors such as different potential transport mechanisms include advection (the movement of containment with leachate flow) and diffusion (the movement of molecules or ions

Table 11. Requirement of A Safe Liner System.[16]

Requirement	Properties that Need to be Checked	Site-Specific Influences
Stability: The liner system should be stable with respect to the mechanical influences without significant change in its leachate behaviors	Shear resistance cohesion (residual/nonresidual values)	Mechanical influences: • Forces resulting from deformation • Forces resulting from overburden loads and inclination • Forces resulting from construction procedures
Imperviousness: Pollution migration through the liner system should be comparable to that for a definable standard size.	(a) Permeability of the liner system hydraulic conductivity and diffusion coefficient retention capacity (b) Sensitivity of the system to imperfections	• Hydraulic gradient • Kind of pollution • Amount of soluble pollutant • Concentration of pollutant in solution • Temperature If a composite liner is considered • Kind of clay • Zone of higher permeability • Deformation or desiccation • Over burden loads
Resistance: If proved that the lining system being exposed to the site-specific influences is still stable and sufficiently impermeable, combination of influences should be considered	Resistance to leachate Resistance to gas Resistance to temperature Hydraulic resistance Resistance to exposure	Chemical influences: • Kind of composition of leachate • Duration of exposure Thermal influences: • Low/high temperature • Duration of exposure Hydraulic influences: • Forces resulting from water movement • Climate and hydrogeology of the site

from high concentration to low concentration region) and the potential attenuation mechanisms include biodegradation, sorption, and dilution.[17]

Diffusion is defined as the movement of molecules or ions due to own random kinetic activity from the areas of higher concentration to the areas of lower concentration. This movement in porous media is given by Fick's law[17] as:

$$f = -n_c D_c \frac{\partial c}{\partial z},$$

where f = mass flux ($\mathrm{ML}^{-2}\,\mathrm{T}^{-1}$), n_c = effective porosity $(-)$, c = concentration in liner (ML^{-3}), and z = distance parallel to diffusion direction.

Most liners such as geomembrane are not conventional porous media, as the pore size is large relative to both water and contaminant molecules. The diffusion of penetrant molecules through a geomembrane, therefore, can be represented by Fick's law as :

$$f = -D_{\mathrm{g}}\frac{\partial c_g}{\partial z},$$

where f = mass flux ($\mathrm{ML^{-2}\ T^{-1}}$), D_{g} = Diffusion coefficient in the liner ($\mathrm{L^2T^{-1}}$), c_{g} = concentration of penetrant in the landfill ($\mathrm{ML^{-3}}$), and z = distance parallel to diffusion direction.

Taking into the conservation mass governing differential equation for transient diffusion as:

$$\frac{\partial c_{\mathrm{g}}}{\partial t} = -D_{\mathrm{g}}\frac{\partial^2 c_{\mathrm{g}}}{\partial z^2}.$$

When a liner is in contact with liquid/leachate for a sufficient time, a final equilibrium that takes place in the liner as per Henry's law[18,19]:

$$c_{\mathrm{g}} = S_{\mathrm{gf}}c_{\mathrm{f}},$$

where c_{g} = final equilibrium concentration in the liner ($\mathrm{ML^{-3}}$), S_{gf} = solubility, partitioning, or Henry's coefficient (–), and c_{f} = equilibrium concentration in the adjacent fluid c_{f} ($\mathrm{ML^{-3}}$).

Alternatively, it can take a nonlinear as Langmuir form

$$c_{\mathrm{g}} = \frac{S_{\mathrm{a}}\mathrm{b}c_{\mathrm{f}}}{1 + \mathrm{b}c_{\mathrm{f}}}$$

where S_a = experimental determined constant and b = variables constant.

Assume that the pollutant permeant is not interact with the liner yield

$$f = -D_{\mathrm{g}}\frac{\partial c_{\mathrm{g}}}{\partial z} = S_{\mathrm{gf}}D_{\mathrm{g}}\frac{\partial c_{\mathrm{f}}}{\partial z} = -P_g\frac{\partial c_{\mathrm{f}}}{\partial z},$$

where

$$P_{\mathrm{g}} = S_{\mathrm{gf}}.$$

P_{g} is thus referred as the permeability which the mass flux across a landfill is given by

$$\begin{aligned} f &= S_{\mathrm{gf}}D_{\mathrm{g}}\frac{\Delta c_{\mathrm{f}}}{t_{\mathrm{GM}}} \\ &= P_{\mathrm{g}}\frac{\Delta c_{\mathrm{f}}}{t_{\mathrm{GM}}}, \end{aligned}$$

where Δc_{f} = difference in concentration in the fluid on either side of the liner.

As transient contaminant transport is controlled by the diffusion coefficient D_{g} instead of the permeability, P_{g}, the Henry coefficient S_{gf} is defined when an equilibrium is reached[20] as:

$$S_{\mathrm{gf}} = \frac{c_{\mathrm{fo}} V_{\mathrm{g}} c_{\mathrm{fF}} (V_{\mathrm{s}} + V_{\mathrm{r}}) - \Sigma V_{\mathrm{i}} c_{\mathrm{i}}}{A t_{\mathrm{GM}} c_{\mathrm{fF}}},$$

where $c_{\mathrm{fo}} =$ initial concentration of fluid in the source reservoir (ML^{-3}), $c_{\mathrm{fF}} =$ final equilibrium concentration in the source and reservoir, V_{s}, $V_{\mathrm{r}} =$ volumes of the source and the reservoir (L^3), $A =$ area of geomembrane through which diffusion occurs (L^2), $t_{\mathrm{GM}} =$ thickness of geomembrane, and $\Sigma V_{\mathrm{i}} c_{\mathrm{i}} =$ mass removed by sampling events (M).

Advection and diffusion are important transport mechanisms of leachate for well-designed and -operated landfills. The rate of advection and diffusion will depend on the level of physical action that occurs in the liner and the level of biodegradation of organic contaminants in the landfill.

Advection is the transport or migration of leachate constituents because of holes or physical defects in the geomembrane liner and flows through the pores of underlying soil layer due to the influence of hydraulic head. Darcy's law can also be used to compute the advective flux for liner.

The advection dispersion, equation with a first-order reaction, is expressed as[21]

$$\frac{\partial c}{\partial t} = D_{xx} \frac{\partial^2 c}{\partial x^2} + D_{xy} \frac{\partial^2 c}{\partial \alpha \partial y} + D_{yy} \frac{\partial^2 c}{\partial y^2} - \vartheta_x \frac{\partial c}{\partial x} - \vartheta_y \frac{\partial c}{\partial y} - kC,$$

where $c =$ dissolved concentration (ML^3), $t =$ time (T), k = first-order reaction rate coefficient (T^{-1}), D_{xx}, D_{yy}, D_{yx}, $D_{xy} =$ dispersion coefficient ($\mathrm{L}^2\mathrm{T}^{-1}$), and $\vartheta_x \vartheta_y =$ velocity component in x and y direction (LT^{-1}).

3. Conclusion

This chapter presents technical guidance for the usage of liners to enhance containment of pollutants in landfill. It is suggested that liners that prevent containment release from landfill to surrounding environment must be properly designed, installed, constructed, and operated. Liners must also be selected from physical, chemical, and biological aspects. Various options of liner including clay liner, various geosynthetic such as geomembrane, geotextile and geonet, GCL, and granular leachate collection system are considered for landfill application to minimize contamination caused to environment. Performance of liners is subject to factors such as waste composition, waste age, and waste decomposition.

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Chapter 6

BENEFICIAL REUSE OF WASTE PRODUCTS

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Abstract
Industry has become an essential part of modern society and waste production is an inevitable outcome of the developmental activities. A material becomes waste when it is discarded without expecting to be compensated for its inherent value. These wastes may pose a potential hazard to the human health or the environment (soil, air, and water) when improperly treated, stored, transported, or disposed off or managed. In this viewpoint, the management of hazardous wastes including their disposal in environment-friendly and economically viable way is very important and, therefore, suggestions are made for developing better strategies. Out of the various categories of the wastes; this chapter introduces different types of waste to be managed using different ways to be environment friendly as wastewater treatment, waste oil management, and finally recycling of waste plastics.

Keywords: Waste management, wastewater, waste oil, biodiesel, recycling, polymers.

1. Waste Management

1.1. *Introduction*

The past few centuries have witnessed the dramatic change from a human population that through of itself powerless to change the earth and anything on its face to a generation that can literally blow the earth up if it so chooses. Simply the slow insidious process of destroying our own home by contaminating and killing the earth's ability to support life is what we call *environmental pollution*. There are two basic reasons for our concern with environmental pollution:

1. Human health and welfare and
2. Our care about the reminder of nature.

Our personal concern with environmental pollution revolves around immediate health problems, and there are plenty of them to worry about from chlorinated pesticides through heavy metals and asbestos to contaminated indoor air. The idea that contamination of the environment is detrimental is not new. Risk due to the contamination of our environment is a ubiquitous and inevitable part of our live. The fact that environmental pollution can make people sick and even prematurely kill people through various forms of environmentally mitigated diseases is a major driving force for environmental pollution control. Risks can happily be reduced significantly by controlling environmental pollution. Human activities naturally produce by-product that are often characterized as "waste." Generally, the more sophisticated the activity is, the more dangerous will be the waste which need more effort to prevent contamination.[1]

Industry has become an essential part of modern society and waste production is an inevitable outcome of the developmental activities. A material becomes waste when it is discarded without expecting to be compensated for its inherent value. These wastes may pose a potential hazard to the human health or the environment (soil, air, and water) when improperly treated, stored, transported, or disposed off or managed.[2]

Supply of water is critical to the survival of life. People, animals, and plants all need water to survive. The concern with water pollution was a concern about health effects. We now think of water pollution not so much in terms of health as in terms of its effects on the aquatic ecosystem and on fish and shellfish in particular. Industries, municipalities, and storm water runoff contribute to the pollution of natural water systems. Factories use water and their industrial wastes are probably the greatest single water pollution problem municipal wastes, agriculture wastes, and pollution from petroleum compounds are another sources of water pollution. One of the main problems with wastewater treatment is that much of the contents of the water are actually solid. The separation of these solids from the water is in fact one of the primary objectives of treatment. The concern with pollution from petroleum compounds is relatively new starting in 1967 with Torrey Canyon disaster; it was the first big spill. In the spring of 1989, a grounded tanker in Alaska's Gulf of Valdez spilled enough oil that complete clean up proved impossible.[1]

Large-scale production of a variety of chemicals and energy and other developmental activities such as agriculture, urbanization, and health care during the past four decades have led to the release of huge quantities of wastes into the environment in the form of solid, liquid, and gases. A substantial amount of these wastes is potentially hazardous to the environment and is extremely dangerous to the living organisms including human beings. There are ample evidence that improper disposal of these wastes may contaminate air (via volatilization and fugitive dust emissions), surface water (from surface runoff or overland flow and groundwater seepage), groundwater (through leaching/infiltration), soils (due to erosion, including fugitive dust generation/deposition and tracking), sediments (from surface runoff/overland flow seepage and leaching), and biota (due to biological uptake and bioaccumulation).

Contamination of groundwater by landfill leachate (a water-based solution of compounds from the waste) posing a risk to downstream surface waters and wells is considered to constitute the major environmental concern associated with the land filling of the waste.[3] In order to safeguard our environment, it is important to regulate such hazardous waste in environmentally feasible and sound manner. The wastes released to air as emanations, naturally are regulated in site, as per legal requirements. For liquid effluent also, it is treated at site or through common legal requirements are fulfilled before release.

The objective of wastewater treatment is to reduce the concentrations of specific pollutants to the level where the discharge of the effluent will not adversely affect the environment. The field of wastewater treatment is littered with unique and imaginative processes for achieving high degree of waste stabilization at attractive costs. However, the problem has been the inattention to the sludge problem. The sources and quantities of sludge depend on various types of wastewater treatment systems.

Solid wastes other than hazardous materials and radioactive wastes are important. Such solid wastes are often called municipal solid waste (MSW) and consist of all the solid and semisolid materials discarded by a community. The fraction of MSW produced by a household is called refuse. Refuse until fairly recently was mostly food wastes, but new materials such as plastics, new packages for products such as cans, and new products such as garbage grinders have all changed the composition MSW. Industry creates thousands new products each year all of which eventually find their way into municipal refuse and contribute to individual disposal problems. The solid waste problem has three facets: source, collection, and disposal. The source of solid waste is perhaps the most difficult of the three to tackle. One potential solution would be to simply redefine solid waste as a resource, and use it for the production of goods for people. Solid wastes often called the "third pollution," which are only now being considered a problem equal in magnitude to that of air and water pollutions. It has been suggested that one solution to the solid waste problem is the development of truly biodegradable forms of materials such as plastics and glass.[1]

However, the situation is more complex with hazardous solid wastes and total storage, disposal, or utilization is not feasible in within the premises on the long run. Hazards waste is relatively new concern of environmental engineers. Such wastes need special collection and treatment and the present write-up covers mostly issues with solid waste from industrial units and sediment and sludge from polluted habitat. The problem of management is complicated by the heterogeneity of the waste in term of chemistry, bioavailability, and toxicity. Even though biomedical and municipal waste are also serious problems, at this junction, it is important to recognize the fact that there are varying degrees of hazards associated with different waste streams and there are good economic advantages for ranking wastes according to the level of hazards they present and also to identify the compatibilities and/or noncompatibilities of the chemical constituents of waste streams for the development of effective waste management program. For instance, waste containing both selenium and mercury would be less likely to create toxicity problems than waste containing only mercury.[4]

Air pollution: interestingly, we have difficulty in defining what is clean air. From the scientific standpoint, clean air is composed of the constituents: nitrogen 780.9, oxygen 209.4, argon 9.3, carbon dioxide 315, neon 18, helium 5.2, methane 1.0–1.2, krypton 1.0, nitrous oxide 0.5, hydrogen 0.5, xenon 0.008, nitrogen dioxide 0.02, and ozone 0.01–0.04 ppm. Therefore, any naturally occurring suspended material can be called a pollutant and one never find such clean air in nature. It may thus be more appropriate to define air pollutants as those substances that exist in such concentration as to cause unwanted effect. Among the gaseous importance are carbon monoxide, ozone, hydrocarbons, hydrogen sulfide, nitrogen oxides and others oxidants, and sulfur oxides. Carbon dioxide may be added to this list because of its potential effect on climate. Perhaps the most familiar effect of air pollution on materials is soiling of building surfaces, clothing, and other articles. Soiling results from the deposition of smoke on surfaces. Over a period of time, this deposition becomes noticeable as soiling, a discoloring, or darkening of the surface. Another effect of air pollution is that of accelerating the corrosion of metals. A second global atmospheric effect of air pollution is the destruction of the ozone layer due to the emission of chlorofluorocarbons (freons). The greatest source of these emissions is from chlorofluorocarbons aerosol propellants. Leaking refrigeration systems are also a source. The heavily polluted air can indeed be a serious threat to our health. The primary objective of air quality management is to attain low air pollutant concentrations in community air. It is possible to reduce the amount of polluted emitted by attaining adequate dispersal and dilution in the atmosphere. Often the easiest solution to air pollution is to stop or change the guilty process. The most serious problem in air pollution is collection of the pollutants so as to provide treatment.[1]

1.2. *Waste*

What is referred to as rubbish, trash, garbage, or junk is unwanted or unusable material. In living organisms, waste is the unwanted substances or toxins that are expelled from them. More commonly, waste refers to the materials that are disposed of in a system of waste management. It is directly linked to human development, both technologically and socially. The composition of different waste has varied over time and location, with industrial development and innovation being directly linked to waste materials. Some components of waste have economical value and can be recycled once correctly recovered. Waste is sometimes a subjective concept, because items that some people discard may have value to others. It is widely recognized that waste materials are a valuable resource, whilst there is debate as to how this value is best realized.

There are many waste types defined by modern systems of waste management, notably including:

- MSW;
- construction and demolition (C&D) wastes;
- institutional waste, commercial waste, and industrial waste (IC&I);
- medical waste (also known as clinical waste);
- hazardous waste, radioactive waste, and electronic waste; and
- biodegradable waste.

Substances or objects that are disposed of or are intended to be disposed of or are required to be disposed of by the provisions of national law can be called waste.[5]

Wastes are materials that are not prime products for which the generator has no further use in terms of own purposes of production, transformation, or consumption, and of which they wants to dispose. Wastes may be generated during the extraction of raw materials, the processing of raw materials into intermediate and final products, the consumption of final products, and other human activities. Residuals recycled or reused at the place of generation are excluded.[6] They define waste as an object the holder discards, intends to discard or is required to discard. Once a substance or object has become waste, it will remain waste until it has been fully recovered and no longer poses a potential threat to the environment or to human health.[7]

1.3. *What is Waste?*

Waste materials are generated from three main sources: (1) residential, (2) institutional, commercial, and industrial (ICI), and (3) C & D.

1.3.1. *Residential Waste*

Solid waste material generated, diverted, or disposed from residential dwellings, primarily private homes. It is estimated that 34% of the solid waste in the province is generated from this source.

1.3.2. *ICI Waste*

Waste material generated, diverted, or disposed of from ICI establishments such as manufacturing, transportation, retail, wholesale and warehousing, commercial (e.g., restaurants and banks), and noncommercial (e.g., health and education) services. This type of waste accounts for approximately 60% of the solid waste in the province.

1.3.3. *C&D Waste*

Solid waste material from residential and commercial construction, renovation, demolition, and land clearing, as well as road and bridge construction waste (concrete and asphalt).

Approximately 6% of the solid waste in the province comes from this source. Most of the waste generated is classified as paper and organic. The remainder is classified as plastic, glass, and inorganic. The data below outlines the percentage of each type of waste generated in the province: paper 37%, organic 30%, metal 9%, plastic 8%, glass 6%, inorganic 4%, and other 6%.[8]

1.4. ***Waste Management***

The primary objective of waste management is to protect the public and environment from potential harm. It is literally the process of managing waste materials. It involves the collection, transport, processing, and/or disposal of waste materials. Historically, the aim of waste management has been to prevent or reduce the impact of waste materials on human health or local amenity. Waste management can involve solid, liquid, and/or gaseous wastes, and the methods involved for each are disparate. Entire fields of expertise exist for the management of each type of waste. Waste management practices are often very different between urban and rural areas, and residential and industrial/commercial producers, even within the same local region. Waste management for nonhazardous residential and institutional waste streams in metropolitan areas is usually (but not always) the responsibility of local government authorities. Waste management for nonhazardous commercial and industrial wastes is usually the responsibility of the generator. The management and composition of waste is usually quite different in developed and developing nations, for a number of reasons.[8]

1.5. ***The Waste Hierarchy***

While waste cannot, in practical terms, be completely eliminated, the amount of waste going for disposal can and must be reduced. The waste hierarchy classifies different waste management strategies according to their desirability. The term “3 Rs” or “Reduce-Reuse-Recycle” has also been used for the same purpose. The waste hierarchy has taken many forms over the past decade, but the basic concept has remained the cornerstone of most waste minimization strategies. The aim of the waste hierarchy is to extract the maximum practical benefits from products and to generate the minimum amount of waste.

Some waste management experts have recently incorporated a “fourth R”: “Re-think,” with the implied meaning that the present system may have fundamental flaws, and that a thoroughly effective system of waste management may need an entirely new way of looking at waste. Some “re-think” solutions may be counter-intuitive, such as cutting fabric patterns with slightly *more* “waste material” left — the now larger scraps are then used for cutting small parts of the pattern, resulting in a decrease in net waste.[7] Successful waste diversion removes volumes of specific types of material from the waste stream and is based on the widely accepted hierarchy of waste management: reduce, reuse, recycle, recovery, and disposal.

1.5.1. *Hierarchy of Pollution and Waste Management Strategies*

- Prevention — This strategy prevents wastes from ever being formed in the first place.
- Recycling/resource recovery/waste-to-energy (R3WE) — Recycling and reuse of materials, the recovery of certain wastes for reuse (known as *resource recovery*), and the conversion of certain types of waste into useful energy such as heat, electricity, and hot water are strategies that recover and offset costs for overall waste management.
- Treatment — When wastes cannot be prevented or minimized through reuse or recycling, then we need to pursue strategies aimed at reducing volumes and/or toxicity. Treatment technologies are processes that focus on stabilization of wastes, reducing toxicity, reducing volume before ultimate disposal, or in some cases creating limited-use by-products.
- Disposal — The only other strategy available is disposal. Waste disposal practices are integrated into the environmental management strategies of all municipalities, are integral to most manufacturing operations, and quite often are among the highest direct cost components. From a business standpoint, it is the least desirable strategy and one that can be directly addressed by waste minimization.[9]

1.5.2. *Hierarchy of Waste Management*

- **Reduce**: Decrease the amount of waste created in the first place; it can be achieved through reduction in use of once-through, disposable, and limited use products, production of more durable goods, and elimination of excess packaging.
- **Reuse**: Reuse materials and products rather than discard them can displace the need for new production; for example, beverage containers, rechargeable batteries, and reusable laser-printer cartridges.
- **Recycle**: Reprocess or recycle waste materials into another usable form displaces the need for new material in production of consumer goods; for example, recycling newsprint, bond paper, and cardboard can displace some of the original fiber used to make new paper products.

- **Recovery**: Gain some useful benefit (materials or energy) from waste; for example, most organic material can be broken down through decomposition to form a rich soil-like material, compost, which can be used to enrich soil or as a cover material on disturbed land.
- **Disposal**: Final placement of waste material with no economic or environmental benefits achieved.[8]

The handling of garbage — solid waste — has become an increasing concern in today's society. Effective solid waste management is a challenge for communities and governments throughout.

Communities are seeing nuisance problems from poorly or improperly maintained landfill sites and concerns have been expressed that the many, unsightly dumps are not in keeping with the expectations of the province's growing tourism industry. Many of the landfill sites experience problems with uncontrolled burning at the sites, smoke, vermin, odors, and excessive wind borne litter.

Waste management is the responsibility of everyone — individuals, communities, businesses, industries, and government. The planning and delivery of waste management is the direct responsibility of municipalities and communities, while the provincial government must provide the framework for waste management by setting policies, regulations, and standards. Government recognizes that it is now time to provide a new framework through the development of a provincial waste management strategy.[8]

1.6. *Waste Management Techniques*

Managing domestic, industrial, and commercial wastes has traditionally consisted of collection, followed by disposal. Depending upon the type of waste and the area, a level of processing may follow collection. This processing may be to reduce the hazard of the waste, recover material for recycling, produce energy from the waste, or reduce it in volume for more efficient disposal. Collection methods vary widely among different countries and regions. Disposal methods also vary widely. In Australia, the most common method of disposal of solid waste is to landfills, because it is a large country with a low-density population. By contrast, in Japan, it is more common for waste to be incinerated, because the country is smaller and land is scarce.[10]

There are many different methods of disposing of waste. Landfill is the most common and probably accounts for more than 90% of the nation's municipal refuse even though landfills have been proven contaminates of drinking water in certain areas. It is the most cost effective method of disposal, with collection and transportation accounting for 75% of the total cost. In a modern landfill, refuse is spread thin, compacted layers covered by a layer of clean earth. Pollution of surface water and groundwater is minimized by lining and contouring the fill, compacting and planting the uppermost cover layer, diverting drainage, and selecting proper soil in sites not subject to flooding or high groundwater levels. The best soil for a landfill is clay because clay is less permeable than other types of soil. Materials disposed of in a landfill can be further secured from leakage by solidifying them in materials such as cement, fly ash from power plants, asphalt, or organic polymers.[11]

1.7. *Disposal Methods*

1.7.1. *Landfill*

Disposing of waste in a landfill is the most traditional method of waste disposal and it remains a common practice in most countries. A well-run landfill can be a hygienic and relatively inexpensive method of disposing of waste materials. Older or poorly managed landfills can create a number of adverse environmental impacts, including wind-blown litter and attraction of vermin and soluble contaminants (leachate) leaching into and polluting groundwater. Another product of landfills containing putrescible wastes is landfill gas (mostly composed of methane and carbon dioxide), which is produced as the waste breaks down. Characteristics of a modern, well-run landfill should include methods to contain leachate, such as clay or plastic liners. Disposed waste should be compacted and covered to prevent vermin and wind-blown litter. Many landfills also have a landfill gas extraction system installed after they are closed to extract the gas generated by the decomposing waste materials. This gas is often burnt to generate power. Generally, even flaring the gas off is a better environmental outcome than allowing it to escape to the atmosphere, as this consumes the methane (a far more potent greenhouse gas than carbon dioxide).

Many local authorities (especially in urban areas) have found it difficult to establish new landfills, due to opposition from adjacent landowners. Few people want a landfill in their local neighborhoods. As a result, solid waste disposal in these areas has become more expensive, as material must be transported further away for disposal.

Some oppose the use of landfills in any way, anywhere, arguing that the logical end result of landfill operations is that it will eventually leave a drastically polluted planet with no canyons, and no wild space. Some futurists have stated that landfills will be the “mines of the future”: as some resources become more scarce, they will become valuable enough that it would be necessary to “mine” them from landfills where these materials were previously discarded as valueless.

Growing concern about the impacts of excessive materials consumption has given rise to efforts to minimize the amount of waste sent to landfill in many areas. These efforts include taxing or levying waste sent to landfill, recycling the materials, converting material to energy, designing products that require less material, etc. A related subject is that of industrial ecology, where the material flows between industries is studied. The by-products of one industry may be a useful commodity to another, leading to reduced waste materials.[10]

1.7.1.1. Common landfill

This disposal method includes mixing of waste with soil, evaporation and infiltration, and/or shallow burial. Solid wastes will normally be incorporated in a landfill and buried. Liquids, slurries, and sludges might also be incorporated into a landfill. The potential for environmental damage by landfilled hazardous wastes differs depending on both the composition and quantity of that waste. Many general-purpose landfills will accept small quantities of hazardous wastes, particularly, if they are in drums or plastic containers. Cyanides, arsenic compounds, and some heavy metal compounds are examples of such materials.

A site which is suitable for the disposal of inert solid wastes and decomposable organic materials. The site must provide separation of the wastes from underlying or adjacent usable water because of leachate possibilities.

The prime requisite for such disposal is that the hazardous contents of the landfill be isolated from the surrounding environment. Water quality of surface and groundwaters must not be compromised. Air quality must also be maintained. Depending upon the nature of waste, various types of secured landfills such as open, control, and closed are used.

Open landfills. Wastes such as metal cans and glass bottles are buried in open landfills.

Controlled landfills. Weakly toxic or potentially toxic wastes, such as incinerator ash or sewage sludges, are buried in controlled landfills.

Closed landfills. Highly toxic wastes such as soils contaminated with polychlorinated biphenyls or mercury compounds are buried in closed landfills where the wastes are completely isolated from the outer environment by a concrete floor, wall, and roof. As rainwater does not enter the closed landfills, the amount of leachate is small.

The major environmental concern with secure landfill is groundwater contamination associated with infiltration of leachate. In order to minimize the infiltration of leachate into the surface soils below the landfill and to substantially reduce the potential for groundwater contamination.[12]

1.7.2. *Incineration*

Incineration is the process of destroying waste material by burning it. Incineration is carried out both on a small scale by individuals and on a large scale by industry. Though still widely used in many areas (especially developing countries), incineration as a waste management tool is becoming controversial for several reasons: both the gases and the ash residue produced may be toxic. Incineration is a controlled process that involves oxidative conversion of combustible solid material to harmless gases suitable for atmospheric release. It converts a waste to a less bulky, less toxic, or less noxious material. The principal products of incineration from a volume standpoint are carbon dioxide, water, and ash while the products of primary concern, due to their environmental effects are compounds containing sulfur, nitrogen, and halogens.[13]

Incineration is often used to produce electricity from waste materials by burning it to produce steam to drive an electrical generator. Energy recovery by incineration is inefficient, however, as even the best incinerator can only recover a fraction of the calorific value of fuel materials. Modern and well-run incinerators employ elaborate pollution control measures on exhaust gases to reduce the amount of toxic products released. Waste gases are of much greater concern in older and/or poorly run incinerators, which can have a significant negative impact on the health of local populations. In recent years, concern has increased about the levels of dioxins that are released when burning mixed waste.

Incinerator ash is toxic and its leachate can pollute groundwater. Until recently, safe disposal of incinerator waste was a major problem. An alternative use for incinerator ash has been to separate it chemically into lye and other useful chemicals. It is also recognized that incineration may be a poor use for many waste materials. Not only is the raw material lost, but also all of the energy and natural resources (such as water) that was used to produce it. Nevertheless, incineration is recognized as a practical method of disposing of hazardous waste materials (such as biological medical waste).[10]

Unless adequate controls are exercised, incineration can lead to atmospheric release of undesired materials. Under such circumstances, a secondary treatment such as after burning, scrubbing, or filtration is required to lower concentrations to acceptable levels prior to atmospheric release. For example, hydrogen chloride generated during incineration of chlorinated hydrocarbons is now being removed through absorption in water. The hydrochloric acid so formed is then either neutralized or concentrated for reuse. This also reduces the risk of dioxin formation. The solid and liquid effluents from the secondary treatment processes will occasionally require treatment prior to ultimate disposal. Incineration reduces the sludge to an ash residue which can be more easily handled for ultimate disposal but at the same time could also lead to the formation of poisonous soluble metal oxides. This concentration in the incinerator ash has to be tackled by recovery/detoxification/disposal.[14]

1.8. *Resource Recovery Techniques*

A relatively recent idea in waste management has been to treat the waste material as a resource to be exploited, instead of simply a challenge to be managed and disposed of. There are a number of different methods by which resources may be extracted from waste: the materials may be extracted and recycled, or the calorific content of the waste may be converted to electricity. The process of extracting resources or value from waste is variously referred to as secondary resource recovery, recycling, and other terms. The practice of treating waste materials as a resource is becoming more common, especially in metropolitan areas where space for new landfills is becoming scarcer. There are a number of methods of recovering resources from waste materials, with new technologies and methods being developed continuously.[10]

1.9. *Recycling*

Recycling means to reuse a material that would otherwise be considered waste. The popular meaning of "recycling" in most developed countries has come to refer to the widespread collection and reuse of single-use beverage containers. These containers are collected and sorted into common groups, so that the raw materials of the items can be used again (recycled). In developed countries, the most common consumer items recycled include aluminum beverage cans, steel food and aerosol cans, HDPE and PET plastic bottles, glass bottles and jars, paperboard cartons, newspapers, magazines, and cardboard. Other types of plastics (polyvinyl chloride (PVC), low-density polyethylene (LDPE), polypropylene (PP), and PS) are also recyclable, although not as commonly collected. These items are usually composed of a single type of material, making them relatively easy to recycle into new products. The recycling of obsolete computers and electronic equipment is important although more costly due to the separation and extraction problems. The recycling of junked automobiles also depends on the scrap metal market. Recycled or used materials have to compete in the marketplace with new (virgin) materials. The cost of collecting and sorting the materials usually means that they are equally or more expensive than virgin materials.

Not accounted for by most economic systems are the benefits to the environment of recycling these materials, compared with extracting virgin materials. It usually requires significantly less energy, water, and other resources to recycle materials than to produce new materials. For example, recycling 1000 kg of aluminum cans saves approximately 5000 kg of bauxite ore being mined and 95% of the energy required to refine it.

In many areas, material for recycling is collected separately from general waste, with dedicated bins and collection vehicles. Other waste management processes recover these materials from general waste streams. This usually results in greater levels of recovery than separate collections of consumer-separated beverage containers, but are more complex and expensive.[10]

1.10. *Composting and Digestion*

Waste materials that are organic in nature, such as food scraps and paper products, are increasingly being recycled. These materials are put through a composting or artificial digestion process to decompose the organic matter and kill pathogens. The organic material is then recycled as mulch or compost for agricultural or landscaping purposes.

There are a large variety of composting methods and technologies, varying in complexity from simple window composting of shredded plant material to automated enclosed-vessel digestion of mixed domestic waste. Composting methods can be broadly categorized into aerobic or anaerobic methods, although hybrids of the two methods also exist. Aerobic (meaning "requiring air") methods of composting seek to aerate the organic material continuously or frequently, in order to promote rapid and odorless decomposition. Anaerobic ("not requiring air") methods of composting seek to maximize the generation of gases such as methane during the process, in order to produce power from the waste materials.[10].

1.11. *Pyrolysis and Gasification*

Pyrolysis and gasification are two related forms of thermal treatment where materials are incinerated with limited oxygen. The process typically occurs in a sealed vessel, under high temperature and pressure. Converting material to energy this way is more efficient than direct incineration, with more energy able to be recovered and used. Pyrolysis of solid waste converts the material into solid, liquid, and gas products. The liquid oil and gas can be burnt to produce energy or refined into other products. The solid residue (char) can be further refined into products such as activated carbon. Gasification is used to convert organic materials directly into a synthetic gas composed of carbon monoxide and hydrogen. The gas is then burnt to produce electricity and steam. Gasification is used in biomass power stations to produce renewable energy and heat.[10]

In this chapter, we choose different topics to clarify:

1. Wastewater treatment,
2. Waste oil management, and
3. Recycling of waste plastic.

2. Wastewater Treatment

Saving water to save the planet and to make the future of mankind safe is what we need now. The fast technological and industrial development, and tumultuous demographic growth and rapid urbanization, especially in the last two decades, are confronting the mankind with four large problems: water, food, energy, and environment. The water problem is particularly pronounced, because it is implicitly present in other three problems, which are in the food and energy production that depend primarily on the water. The demand for water ("Water for People Water for Life" United Nations World Water Development Report UNESCO) has increased tremendously with agricultural, industrial, and domestic sectors consuming 70%, 22%, and 8% of the available freshwater, respectively and this has resulted in the generation of large amounts of wastewater.[15–17] Therefore, freshwater is a vital natural resource that will continue to be renewable as long as it is well managed. Preventing pollution from municipal wastewater that includes domestic, industrial, and agro-industrial activities is important to ensure the sustainability of the locales development. Undoubtedly, the water pollution control efforts that have been underway in many countries have already achieved some success. Nevertheless, the problems that are confronted grow in complexity and intensity. The pollution of freshwater bodies with the consequent deterioration in water quality can only worsen the situation. Such pollution has been brought about by the discharge of inadequately treated sewage and industrial wastewaters. Perhaps not unexpectedly, as the demand for more water is met, the volumes of wastewater can also be expected to increase. Industrial wastewaters (including agro-industrial wastewaters) are effluents that result from human activities that are associated with raw-material processing and manufacturing. These wastewater streams arise from washing, cooking, cooling, heating, extraction, reaction by-products, separation, conveyance, and quality control resulting in product rejection. Water pollution occurs when potential pollutants in these streams reach certain amounts causing undesired alterations to a receiving water body. While industrial wastewaters from such processing or manufacturing sites may include some domestic sewage, the latter is not the major component. Domestic sewage may be present because of washrooms and hostels provided for workers at the processing or manufacturing facility. Examples of industrial wastewaters include those arising from chemical, pharmaceutical, electrochemical, electronics, petrochemical, and food-processing industries. Examples of agro-industrial wastewaters include those arising from industrial-scale animal husbandry, slaughterhouses, fisheries, and seed oil processing. Agro-industrial wastewaters can be very strong in terms of pollutant concentrations and hence can contribute significantly to the overall pollution load imposed on the environment.[18]

2.1. *Importance of Wastewater Treatment*

All major terrestrial biota, ecosystems, and humans depend on freshwater (i.e. water with less than $100\,\mathrm{mg\,L^{-1}}$ salts) for their survival. The earth's water is primarily saline in nature (about 97%). Of the remaining (3%) water, 87% of it is locked in the polar caps and glaciers. This would mean only 0.4% of all water on earth is accessible freshwater. The latter is, however, a continually renewable resource although natural supplies are limited by the amounts that move through the natural water cycle. Freshwater shortages increase the risk of conflict, public health problems, reduction in food production, inhibition of industrial production expansion, and these problems threaten the environment.

Freshwater shortages are, however, not only due to uneven distribution of freshwater resources and demand for freshwater but also, increasingly, due to the declining water quality in freshwater sources already in use. This declining water quality is primarily due to pollution. It should not be forgotten that in the wider context of resources associated with water, the marine environment is also included in the picture. While the latter was, in the past, primarily associated with the fisheries resource, it can also include tourism and the feed for desalination in the current context. Untreated industrial wastewaters would add pollutants into water bodies — freshwater and saline. These receiving water bodies, freshwater and marine, can include ponds, lakes, rivers, coastal waters, and the sea. It would be useful to bear in mind that pollutants introduced into a river or some other freshwater water body do eventually end up in the sea, the ultimate receptacle for waterborne pollutants if these are permitted to find their way through the environment unimpeded. In the last decade, the emergence of the pollutants presence in the water environment can be summarized in the following broad categories:

2.1.1. *Physical Effects*

These include impact on clarity of the water and interference to oxygen dissolution in it. Water clarity is affected by turbidity that may be caused by inorganic (fixed suspended solids (FSS)) and/or organic particulates suspended in the water (volatile suspended solids (VSS)). The latter may undergo biodegradation and thereby also have oxidation effects. Turbidity reduces light penetration and this reduces photosynthesis while the attendant loss in clarity, among other things, would adversely affect the food gathering capacity of aquatic animals because these may not be able to see their prey. Very fine particulates may also clog the gill surfaces of fishes and thereby affecting respiration and eventually killing them. Settleable particulates may accumulate on plant foliage and bed of the water-body forming sludge layers that would eventually smother benthic organisms. As the sludge layers accumulate, they may eventually become sludge banks and if the material in these is organic then its decomposition would give rise to malodors. In contrast to the settleable material, particulates lighter than water eventually float to the surface and form a scum layer. The latter also interferes with the passage of light and oxygen dissolution. Because of the former, these scum layers affect photosynthesis. Discharge limits on wastewater or treated wastewater discharges typically have a value for TSS such as 30 mg L^{-1} or 50 mg L^{-1}. Many industrial wastewaters contain oil and grease (O&G). While some of the latter may be organic in nature, there are many which mineral oils are. Notwithstanding their organic or mineral nature, both types cause interference at the air–water interface and inhibit the transfer of oxygen. Apart from their interference to the transfer of oxygen from atmosphere to water, the O&G (particularly the mineral oils) may also be inhibitory. Unlike domestic sewage, industrial discharges can have temperatures substantially above ambient temperatures. These raise the temperatures of the receiving water and reduce the solubility of oxygen. Apart from this, rapid changes in temperature may result in thermal shock and this may be lethal to the more sensitive species. Heat, however, does not always have a negative impact on organisms, as it may positively affect growth rates although there are limits here too since the condition may favor certain species within the population more than others and over time biodiversity may be negatively affected.

2.1.2. *Oxidation and Residual Dissolved Oxygen* (*DO*)

As suggested in the preceding paragraph, water bodies have the capacity to oxygenate themselves through dissolution of oxygen from the atmosphere and photosynthetic activity by aquatic plants. Of the latter, algae often play an important role. However, there is a finite capacity to this reoxygenation and if oxygen depletion, as a result of biological or chemical processes induced by the presence of organic or inorganic substances that exert an oxygen demand (i.e. as indicated by the biochemical oxygen demand (BOD) or chemical oxygen demand (COD), exceeded this capacity then the DO levels would decline. The latter may eventually decline to such an extent that septic conditions occur. A manifestation of such conditions would be the presence of malodors released by facultative and anaerobic organisms. An example of this is the reduction of substances with combined oxygen such as sulfates by facultative bacteria and resulting in the release of hydrogen sulfide. The depletion of free oxygen would affect the survival of aerobic organisms. DO levels do not; however, need to drop to zero before adverse impacts are felt. A decline to 3–4 mg L^{-1}, which still means the water contains substantial quantities of oxygen, may already adversely affect higher organisms. If inhibitory substances are also present, then the DO level at which adverse effects may be felt can be even higher than before. The case of elevated water temperatures due to warm discharges is somewhat different. The elevated temperatures can affect metabolic rates positively (possibly twofold for each 10°C rise in temperature) but elevated temperatures also reduce the solubility of oxygen in water. This would mean increasing demand for oxygen while its availability declines. Because of the impact of DO levels on aquatic life, much importance has been placed on determining the BOD value of a discharge. Typical BOD_5 limits set are values such as 20 and 50 mg L^{-1}.

2.1.3. *Inhibition or Toxicity and Persistence*

These effects may be caused by organic or inorganic substances and can be acute or chronic. Examples of these include the pesticides and heavy metals mentioned in the preceding section. Many industrial wastewaters do contain such potentially inhibitory or toxic substances. The presence of such substances in an ecosystem may bias a population toward members of the community that are more tolerant to the substances while eliminating those that are less tolerant and resulting in a loss of biodiversity. For similar reasons, an awareness of the impact such substances have on biological systems is not only relevant in terms of protection of the environment but is of no less importance in terms of their impact on the biological systems used to treat industrial wastewaters. Even successful treatment of such a wastewater may not necessarily mean that the potability of water in a receiving water body would not be affected. For example, small quantities of residual phenol in water can react with chlorine during the potable water treatment process giving rise to chlorophenols that can cause objectionable tastes and odors in the treated water. Apart from the organic pollutants that are potentially inhibitory or toxic, there are those that are resistant to biological degradation. Such persistent compounds can be bioaccumulated in organisms resulting in concentrations in tissues being significantly higher than concentrations in the environment and thereby making these organisms unsuitable as prey/food for organisms (including man) higher up the food chain. While some organic compounds may be persistent, metals are practically nondegradable in the environment.

2.1.4. *Eutrophication*

The discharge of nitrogenous and phosphorous compounds into receiving water bodies may alter their fertility. Enhanced fertility can lead to excessive plant growth. The latter may include algal growth. The subsequent impact of such growth on a water body can include increased turbidity, oxygen depletion, and toxicity issues. Algal growth in unpolluted water bodies is usually limited because the water is nutrient limiting. While nutrients would include marconutrients such as nitrogen, phosphorous, and carbon and micronutrients such as cobalt, manganese, calcium, potassium, magnesium, copper, and iron that are required only in very small quantities, the focus in concerns over eutrophication would be on phosphorous and nitrogen as quantities of the other nutrients in the natural environment are often inherently adequate. In freshwaters, the limiting nutrient is usually phosphorous; while in estuarine and marine waters, it would be nitrogen. Treatment of industrial wastewater (or domestic sewage for that matter) can then target the removal of either phosphorous or nitrogen, depending on the receiving water body, to ensure that the nutrient-limiting condition is maintained. Apart from esthetic issues, such algal blooms may affect the productivity of the fisheries in the locale. It should be noted that not all industrial wastewaters contain excessive quantities of macro- and micro-nutrients. This deficiency, if there is, results in process instability and/or the proliferation of inappropriate microbial species during biological treatment of the wastewaters. Bulking sludge is a manifestation of such an occurrence. To address this deficiency, nutrients supplementation is required. The quantities used should be carefully regulated so that an excessive nutrients condition is not inadvertently created and these excess nutrients subsequently discharged with the treated effluent. In terms of BOD:N:P, the optimal ratio for biotreatment is often taken as 100:5:1 while the minimum acceptable condition can be 150:5:1.

2.1.5. *Pathogenic Effects*

Pathogens are disease-causing organisms and infection occurs when these organisms enter into a host (e.g. man or animal) and multiply therein. These pathogens include bacteria, viruses, protozoa, and helminthes. While domestic and medical-related wastewaters may typically be linked to such microorganisms (and especially the bacteria and viruses), industrial wastewaters are not typically associated with this category of effects. The exception to this is wastewaters associated with the sectors in the agroindustry dealing with animals. The concern here would be the presence of such organisms in the wastewater that is discharged into a receiving water body and diseases, if any, are then transmitted through the water. While many of these organisms can be satisfactorily addressed with adequate disinfection of the treated effluent and raw potable water supplies during the water treatment process, there are those that cannot be dealt with so easily. The difficulty is that the infected host does not necessarily shed the organism but is likely also to shed its eggs or oocysts. The latter can unfortunately be resistant to the usual disinfection processes. An outbreak of cryptosporidiosis, a gastrointestinal disease, would result in the hosts suffering from diarrhea, abdominal pain, nausea, and vomiting.[18,19]

2.2. *Characteristics of Wastewater*

An understanding of the nature of wastewater is fundamental for the design of appropriate wastewater treatment plants and the selection of effective treatment technologies. Wastewater originates predominantly from water usage by residences and commercial and industrial establishments, together with groundwater, surface water, and storm water. Consequently, wastewater flow fluctuates with variations in water usage, which is affected by a multitude of factors including climate, community size, living standards, dependability and quality of water supply, water conservation requirements or practices, and the extent of meter services, in addition to the degree of industrialization, cost of water, and supply pressure. Wide variations in wastewater flow rates may thus be expected to occur within a community.[20]

Wastewater quality may be defined by its physical, chemical, and biological characteristics. Physical parameters include color, odor, temperature, and turbidity. Insoluble contents, such as solids, oil, and grease, also fall into this category. Solids may be further subdivided into suspended and dissolved solids as well as organic (volatile) and inorganic (fixed) fractions.

Chemical parameters associated with the organic content of wastewater include BOD, chemical oxygen demand (COD), total organic carbon (TOC), and total oxygen demand (TOD). Inorganic chemical parameters include salinity, hardness, pH, acidity and alkalinity, as well as concentrations of ionized metals such as iron and manganese and anionic entities such as chlorides, sulfates, sulfides, nitrates, and phosphates. Bacteriological parameters include coliforms, fecal coliforms, specific pathogens, and viruses. Both constituents and concentrations vary with time and local conditions. Wastewater is classified as strong, medium or weak, depending on its contaminant concentration. The effects of the discharge of untreated wastewater into the environment are manifold and depend on the types and concentrations of pollutants.[21]

2.3. *Wastewater Treatment Technologies*

Satisfactory disposal of wastewater, whether by surface method, subsurface method, or dilution, is dependent on its treatment prior to disposal. Adequate treatment is necessary to prevent contamination of receiving waters to a degree which might interfere with their best or intended use, whether it be for water supply, recreation, or any other required purpose. Wastewater treatment consists of applying known technology to improve or upgrade the quality of a wastewater. Usually wastewater treatment will involve collecting the wastewater in a central, segregated location (the wastewater treatment plant), and subjecting the wastewater to various treatment processes. Most often, since large volumes of wastewater are involved, treatment processes are carried out on continuously flowing wastewaters (continuous flow or "open" systems) rather than as "batch" or a series of periodic treatment processes in which treatment is carried out on parcels or "batches" of wastewaters. While most wastewater treatment processes are continuous flow, certain operations, such as vacuum filtration, involving as it does storage of sludge, the addition of chemicals, filtration, and removal or disposal of the treated sludge, are routinely handled as periodic batch operations.

Normally wastewater treatment processes[22] consist of following steps like: pretreatment — industrial wastewater streams prior to discharge to municipal sewerage systems or even to a central industrial sewerage system are pretreated doing equalization, neutralization; then they undergo primary treatment and wastewater is directed toward removal of pollutants with the least effort. Suspended solids are removed by either physical or chemical separation techniques and handled as concentrated solids; then they are given a secondary treatment usually involving microorganisms (biological treatment) primarily bacteria that stabilize the waste components. The third step is physicochemical treatment or tertiary treatment and the processes included in this are adsorption, ion exchange, stripping, chemical oxidation, and membrane separations. All of these are more expensive than biological treatment but are used for the removal of pollutants that are not easily removed by biological methods. Though these are generally utilized in series with biological treatment, sometimes they are used as stand-alone processes too. The final step being the sludge processing and disposal. Municipal wastewater was treated using more or less one of these treatment methods; nevertheless, there is no single standard methodology/treatment procedure used for all types of wastes. The general methodologies that used for wastewater treatment are broadly classifiable into three categories:

(i) physical,
(ii) chemical, and
(iii) biological processes.

Some of the methodologies lying in above-mentioned categories are discussed in brief in subsequent sections (Fig. 1).

2.3.1. *Physical Unit Operations*

Among the first treatment methods used were physical unit operations, in which physical forces are applied to remove contaminants. Today, they still form the basis of most process flow systems for wastewater treatment. This section briefly discusses the most commonly used physical unit operations.

2.3.1.1. Screening

The screening of wastewater, one of the oldest treatment methods, removes gross pollutants from the waste stream to protect downstream equipment from damage, avoid interference with plant operations and prevent objectionable floating material from entering the primary settling tanks. Screening devices may consist of parallel

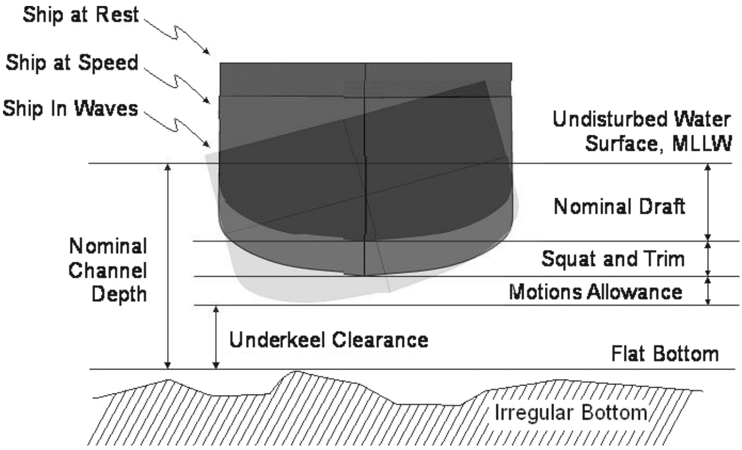


Figure 1. Lists of some of the unit operations included within each category.

bars, rods or wires, grating, wire mesh, or perforated plates, to intercept large floating or suspended material. The openings may be of any shape, but are generally circular or rectangular.[23] The material retained from the manual or mechanical cleaning of bar racks and screens is referred to as "screenings" and is either disposed of by burial or incineration or returned into the waste flow after grinding.[24]

2.3.1.2. Comminution

Comminutors are used to pulverize large floating material in the waste flow. They are installed where the handling of screenings would be impractical, generally between the grit chamber and the primary settling tanks. Their use reduces odors, flies, and unsightliness. A comminutor may have either rotating or oscillating cutters. Rotating-cutter comminutors either engage a separate stationary screen alongside the cutters or a combined screen and cutter rotating together. A different type of comminutor, known as a barminutor, involves a combination of a bar screen and rotating cutters.[25]

2.3.1.3. Flow equalization

Flow equalization is a technique used to improve the effectiveness of secondary and advanced wastewater treatment processes by leveling out operation parameters such as flow, pollutant levels, and temperature over a period of time. Variations are damped until a near-constant flow rate is achieved, minimizing the downstream effects of these parameters.[25] Flow equalization may be applied at a number of locations within a wastewater treatment plant, e.g. near the head end of the treatment works, prior to discharge into a water body, and prior to advanced waste treatment operations. There are four basic flow equalization processes which are:

(i) *Alternating flow diversion*: two basins alternating between filling and discharging for successive time periods.
(ii) *Intermittent flow diversion*: an equalization basin to which a significant increase in flow is diverted. The diverted flow is then fed into the system at controlled rate.
(iii) *Completely mixed and combined flow*: a basin that completely mixes multiple flows at the front end of the treatment process.
(iv) *Completely mixed and mixed flow*: a large, completely mixed, holding basin located before the wastewater facility, leveling parameters in influent stream and providing a constant discharge.[26]

2.3.1.4. Sedimentation

Sedimentation, a fundamental and widely used unit operation in wastewater treatment, involves the gravitational settling of heavy particles suspended in a mixture. This process is used for the removal of grit, particulate matter in the primary settling basin, biological floc in the activated sludge settling basin, and chemical flow when the chemical coagulation process is used. Sedimentation takes place in a settling tank, also referred to as a clarifier. There are three main designs, namely, horizontal flow, solids contact, and inclined surface.[21] In designing a sedimentation basin, it is important to bear in mind that the system must produce both a clarified effluent and a concentrated sludge. Four types of settling occur, depending on particle concentration: discrete, flocculent, hindered, and compression. It is common for more than one type of settling to occur during a sedimentation operation.

2.3.1.5. Flotation

Flotation is a unit operation used to remove solid or liquid particles from a liquid phase by introducing a fine gas, usually air bubbles. The gas bubbles either adhere to the liquid or are trapped in the particle structure of the suspended solids, raising the buoyant force of the combined particle and gas bubbles. Particles that have a higher density than the liquid can thus be made to rise. In wastewater treatment, flotation is used mainly to remove suspended matter and to concentrate biological sludge. The chief advantage of flotation over sedimentation is that very small or light particles can be removed more completely and in a shorter time. Once the particles have been floated to the surface, they can be skimmed out. Flotation, as currently practiced in municipal wastewater treatment, uses air exclusively as the floating agent. Furthermore, various chemical additives can be introduced to enhance the removal process.[21] There are four different flotation methods that are commonly used:

(i) *Dissolved-air flotation*: The injection of air while wastewater is under the pressure of several atmospheres. After a short holding time, the pressure is restored to atmospheric level, allowing the air to be released as minute bubbles.
(ii) *Air flotation*: The introduction of gas into the liquid phase directly by means of a revolving impeller or through diffusers, at atmospheric pressure.
(iii) *Vacuum flotation*: The saturation of wastewater with air either directly in an aeration tank or by permitting air to enter on the suction side of a wastewater pump. A partial vacuum is applied, causing the dissolved air to come out of solution as minute bubbles that rise with the attached solids to the surface, where they form a scum blanket. The scum is removed by a skimming mechanism while the settled grit is raked to a central sump for removal.
(iv) *Chemical additives*: Chemicals further the flotation process by creating a surface that can easily adsorb or entrap air bubbles. Inorganic chemicals (aluminum and ferric salts and activated silica) and various organic polymers can be used for this purpose.

2.3.1.6. Granular medium filtration

The filtration of effluents from wastewater treatment processes is a relatively recent practice, but has come to be widely used for the supplemental removal of suspended solids from wastewater effluents of biological and chemical treatment processes, in addition to the removal of chemically precipitated phosphorus. The complete filtration operation comprises two phases: filtration and cleaning or backwashing. The wastewater to be filtered is passed through a filter bed consisting of granular material (sand, anthracite, and/or garnet), with or without added chemicals. Within the filter bed, suspended solids contained in the wastewater are removed by means of a complex process involving one or more removal mechanisms such as straining, interception, impaction, sedimentation, flocculation, and adsorption. The phenomena that occur during the filtration phase are basically the same for all types of filters used for wastewater filtration. The cleaning/backwashing phase differs depending on whether the filter operation is continuous or semicontinuous. In semi-continuous filtration, the filtering and cleaning operations occur sequentially, whereas in continuous filtration the filtering and cleaning operations occur simultaneously.[21]

2.3.2. *Biological Unit Processes*

Biological unit processes are used to convert the finely divided and dissolved organic matter in wastewater into flocculent settleable organic and inorganic solids. In these processes, microorganisms, particularly bacteria, convert the colloidal and dissolved carbonaceous organic matter into various gases and into cell tissue which is then removed in sedimentation tanks. Biological processes are usually used in conjunction with physical and chemical processes, with the main objective of reducing the organic content (measured as BOD, TOC, or COD) and nutrient content (notably nitrogen and phosphorus) of wastewater. Biological processes used for wastewater treatment may be classified under five major types:

(a) aerobic processes,
(b) anoxic processes,
(c) anaerobic processes,
(d) combined processes, and
(e) pond processes.

These processes are further subdivided, depending on whether the treatment takes place in a suspended-growth system an attached-growth system or a combination of both. This section will be concerned with the most commonly used biological processes, including trickling filters, the activated sludge process, aerated lagoons, rotating biological contactors (RBCs) and stabilization ponds.[25]

2.3.2.1. Activated-sludge process

The activated-sludge process is an aerobic and continuous-flow system containing a mass of activated microorganisms that are capable of stabilizing organic matter. The process consists of delivering clarified wastewater, after primary settling, into an aeration basin where it is mixed with an active mass of microorganisms, mainly bacteria and protozoa, which aerobically degrade organic matter into carbon dioxide, water, new cells, and other end products. The bacteria involved in activated sludge systems are primarily Gram-negative species, including carbon oxidizers, nitrogen oxidizers, floc formers and nonfloc formers, and aerobes and facultative anaerobes. The protozoa, for their part, include flagellates, amoebas, and ciliates. An aerobic environment is maintained in the basin by means of diffused or mechanical aeration, which also serves to keep the contents of the reactor (or mixed liquor) completely mixed. After a specific retention time, the mixed liquor passes into the secondary clarifier, where the sludge is allowed to settle and a clarified effluent is produced for discharge. The process recycles a portion of the settled sludge back to the aeration basin to maintain the required activated sludge concentration. The process also intentionally wastes a portion of the settled sludge to maintain the required solids retention time (SRT) for effective organic removal.

Control of the activated-sludge process is important to maintain a high treatment performance level under a wide range of operating conditions. The principal factors in process control are the following:

(a) Maintenance of DO levels in the aeration tanks;
(b) Regulation of the amount of returning activated sludge; and
(c) Control of the waste-activated sludge.

The main operational problem encountered in a system of this kind is sludge bulking, which can be caused by the absence of phosphorus, nitrogen, and trace elements and wide fluctuations in pH, temperature, and DO. Bulky sludge has poor settleability and compatibility due to the excessive growth of filamentous microorganisms. This problem can be controlled by chlorination of the return sludge.[21]

2.3.2.2. Aerated lagoons

An aerated lagoon is a basin between 1 and 4 meters in depth in which wastewater is treated either on a flow-through basis or with solids recycling. The microbiology involved in this process is similar to that of the activated-sludge process. However, differences arise because the large surface area of a lagoon may cause more temperature effects than are ordinarily encountered in conventional activated-sludge processes. Wastewater is oxygenated by surface, turbine, or diffused aeration. The turbulence created by aeration is used to keep the contents of the basin in suspension. Depending on the retention time, aerated lagoon effluent contains approximately one-third to one-half the incoming BOD value in the form of cellular mass. Most of these solids must be removed in a settling basin before final effluent discharge.

2.3.2.3. Trickling filters

The trickling filter is the most commonly encountered aerobic attached-growth biological treatment process used for the removal of organic matter from wastewater. It consists of a bed of highly permeable medium to which organisms are attached, forming a biological slime layer, and through which wastewater is percolated. The filter medium usually consists of rock or plastic packing material. The organic material present in the wastewater is degraded by adsorption on to the biological slime layer. In the outer portion of that layer, it is degraded by aerobic microorganisms. As the microorganisms grow, the thickness of the slime layer increases and the oxygen is depleted before it has penetrated the full depth of the slime layer. An anaerobic environment is thus established near the surface of the filter medium.

As the slime layer increases in thickness, the organic matter is degraded before it reaches the microorganisms near the surface of the medium. Deprived of their external organic source of nourishment, these microorganisms die and are washed off by the flowing liquid. A new slime layer grows in their place. This phenomenon is referred to as “sloughing”.[26] After passing through the filter, the treated liquid is collected in an under-drain system, together with any biological solids that have become detached from the medium. The collected liquid then passes to a settling tank where the solids are separated from the treated wastewater. A portion of the liquid collected in the under drain system or the settled effluent is recycled to dilute the strength of the incoming wastewater and to maintain the biological slime layer in moist condition.

2.3.2.4. Rotating biological contactors

An RBC is an attached-growth biological process that consists of one or more basins in which large closely spaced circular disks mounted on horizontal shafts rotate slowly through wastewater. The disks, which are made of high-density PS or PVC, are partially submerged in the wastewater, so that a bacterial slime layer forms on their wetted surfaces. As the disks rotate, the bacteria are exposed alternately to wastewater, from which they adsorb organic matter, and to air, from which they absorb oxygen. The rotary movement also allows excess bacteria to be removed from the surfaces of the disks and maintains a suspension of sloughed biological solids. A final clarifier is needed to remove sloughed solids. Organic matter is degraded by means of mechanisms similar to those operating in the trickling filters process. Partially submerged RBCs are used for carbonaceous BOD removal, combined carbon oxidation and nitrification, and nitrification of secondary effluents. Completely submerged RBCs are used for denitrification.[21]

2.3.2.5. Stabilization ponds

A stabilization pond is a relatively shallow body of wastewater contained in an earthen basin, using a completely mixed biological process without solids return. Mixing may be either natural (wind, heat, or fermentation) or induced (mechanical or diffused aeration). Stabilization ponds are usually classified based on the nature of the biological activity that takes place in them, as aerobic, anaerobic, or aerobic anaerobic. Aerobic ponds are used primarily for the treatment of soluble organic wastes and effluents from wastewater treatment plants. Aerobic-anaerobic (facultative) ponds are the most common type and have been used to treat domestic wastewater and a wide variety of industrial wastes. Anaerobic ponds, for their part, are particularly effective in bringing about rapid stabilization of strong concentrators of organic wastes. Aerobic and facultative ponds are biologically complex. The bacterial population oxidizes organic matter, producing ammonia, carbon dioxide, sulfates, water, and other end products, which are subsequently used by algae during daylight to produce oxygen. Bacteria then use this supplemental oxygen and the oxygen provided by wind action to break down the remaining organic matter. Wastewater retention time ranges between 30 and 120 days. This is a treatment process that is very commonly found in rural areas because of its low construction and operating costs.[26]

2.3.2.6. Completely mixed anaerobic digestion

Anaerobic digestion involves the biological conversion of organic and inorganic matter in the absence of molecular oxygen to a variety of end-products including methane and carbon dioxide. A consortium of anaerobic organisms work together to degrade the organic sludges and wastes in three steps, consisting of hydrolysis of high-molecular-mass compounds, acidogenesis, and methanogenesis. The process takes place in an airtight reactor. Sludge is introduced continuously or intermittently and retained in the reactor for varying periods of time. After withdrawal from the reactor, whether continuous or intermittent, the stabilized sludge is reduced in organic and pathogen content and is nonputrescible. The two most widely used types of anaerobic digesters are standard-rate and high-rate. In the standard-rate digestion process, the contents of the digester are usually unheated and unmixed, and are retained for a period ranging from 30 to 60 days. In the high-rate digestion process, the contents of the digester are heated and mixed completely, and are retained, typically, for a period of 15 days or less. A combination of these two basic processes is known as the two-stage process, and is used to separate the digested solids from the supernatant liquor. However, additional digestion and gas production may occur.

Anaerobic digesters are commonly used for the treatment of sludge and wastewaters with high organic content. The disadvantages and advantages of a system of this kind, as compared to aerobic treatment, stem directly from the slow growth rate of methanogenic bacteria. A slow growth rate requires a relatively long retention time in the digester for adequate waste stabilization to occur; however, that same slow growth means that only a small portion of the degradable organic matter is synthesized into new cells. Another advantage of this type of system is the production of methane gas, which can be used as a fuel source, if produced in sufficient quantities. Furthermore, the system produces a well-stabilized sludge, which can be safely disposed of in a sanitary landfill after drying or dewatering. On the other hand, the fact that high temperatures are required for adequate treatment is a major drawback.[26]

2.3.2.7. Biological nutrient removal

Nitrogen and phosphorus are the principal nutrients of concern in wastewater discharges. Discharges containing nitrogen and phosphorus may accelerate the eutrophication of lakes and reservoirs and stimulate the growth of algae and rooted aquatic plants in shallow streams. Significant concentrations of nitrogen may have other adverse effects as well: depletion of DO in receiving waters, toxicity to aquatic life, and adverse impact on chlorine disinfection efficiency, creation of a public health hazard, and wastewater that is less suitable for reuse. Nitrogen and phosphorus can be removed by physical, chemical, and biological methods. Biological removal of these nutrients is described briefly in the following sections.

2.3.2.7.1. *Nitrification-denitrification*

Nitrification is the first step in the removal of nitrogen by means of this process. Biological nitrification is the work of two bacterial genera: *Nitrosomonas*, which oxidizes ammonia to the intermediate product nitrite, and *Nitrobacter*, which converts nitrite to nitrate. Nitrifying bacteria are sensitive organisms and are extremely susceptible to a wide variety of inhibitors such as high concentrations of ammonia and nitrous acid, low DO levels (<1 mg/L), pH outside the optimal range (7.5–8.6), and so on. Nitrification can be achieved through both suspended-growth and attached-growth processes. In suspended-growth processes, nitrification is brought about either in the same reactor that is used for carbonaceous BOD removal or in a separate suspended-growth reactor following a conventional activated sludge treatment process. Ammonia is oxidized to nitrate with either air or high-purity oxygen. Similarly, nitrification in an attached-growth system may be brought about either in the same attached-growth reactor that is used for carbonaceous BOD removal or in a separate reactor. Trickling filters, RBCs, and packed towers can be used for nitrifying systems.[25]

2.3.2.7.2. *Phosphorus removal*

Phosphorus appears in water as orthophosphate (PO_4^{-3}), polyphosphate (P_2O_7), and organically bound phosphorus. Microbes utilize phosphorus during cell synthesis and energy transport. As a result, 10–30% of all influent phosphorus is removed during secondary biological treatment. More phosphorus can be removed if one of a number of specially developed biological phosphorus removal processes is used. Typical biological processes used for phosphorus removal are the proprietary A/O process. The A/O process is a single-sludge suspended-growth system that combines aerobic and anaerobic sections in sequence. Settled sludge is returned to the influent end of the reactor and mixed with the incoming wastewater. In the PhoStrip process, a portion of the return activated sludge from the secondary biological treatment process is diverted to an anaerobic phosphorus stripping tank. There, phosphorus is released into the supernatant, which is subsequently treated with lime or some other coagulant. The phosphorus-poor activated sludge is returned to the aeration tank. These processes are based on the exposure of microbes in an activated-sludge system to alternating anaerobic and aerobic conditions. This stresses the microorganisms, so that their uptake of phosphorus exceeds normal levels.[25]

2.3.3. *Chemical Unit Processes*

Chemical processes used in wastewater treatment are designed to bring about some form of change by means of chemical reactions. They are always used in conjunction with physical unit operations and biological processes. In general, chemical unit processes have an inherent disadvantage compared to physical operations in that they are additive processes. That is to say, there is usually a net increase in the dissolved constituents of the wastewater. This can be a significant factor if the wastewater is to be reused. This section discusses the main chemical unit processes.

2.3.3.1. Chemical precipitation

Chemical coagulation of raw wastewater before sedimentation promotes the flocculation of finely divided solids into more readily settleable flocs, thereby enhancing the efficiency of suspended solid, BOD_5, and phosphorus removal. The degree of clarification obtained depends on the quantity of chemicals used and the care with which the process is controlled.[25] Coagulant selection for enhanced sedimentation is based on performance, reliability, and cost. Performance evaluation uses jar tests of the actual wastewater to determine dosages and effectiveness. Chemical coagulants that are commonly used in wastewater treatment include alum ($Al_2(SO_4)_3 \cdot 14.3\,H_2O$), ferric chloride ($FeCl_3 \cdot 6H_2O$), ferric sulfate ($Fe_2(SO_4)_3$), ferrous sulfate ($FeSO_4 \cdot 7H_2O$), and lime ($Ca(OH)_2$). Organic polyelectrolytes are sometimes used as flocculation aids.[24] Suspended solids removal through chemical treatment involves a series of three unit operations: rapid mixing, flocculation, and settling. First, the chemical is added and completely dispersed throughout the wastewater by rapid mixing for 20–30 s in a basin with a turbine mixer. Coagulated particles are then brought together via flocculation by mechanically inducing velocity gradients within the liquid. Flocculation takes 15–30 min in a basin containing turbine or paddle-type mixers.[26] The final step is clarification by gravity. The advantages of coagulation include greater removal efficiency, the feasibility of using higher overflow rates, and more consistent performance. On the other hand, coagulation results in a larger mass of primary sludge that is often more difficult to thicken and dewater. It also entails higher operational costs and demands greater attention on the part of the operator.

2.3.3.2. Disinfection

Disinfection refers to the selective destruction of disease-causing microorganisms. This process is of importance in wastewater treatment owing to the nature of wastewater, which harbors a number of human enteric organisms that are associated with various waterborne diseases. Commonly used means of disinfection include the following:

- Physical agents such as heat and light;
- Mechanical means such as screening, sedimentation, and filtration;
- Radiation, mainly gamma rays; and
- Chemical agents including chlorine and its compounds, bromine, iodine, ozone, phenol and phenolic compounds, alcohols, heavy metals, dyes, soaps and synthetic detergents, quaternary ammonium compounds, hydrogen peroxide, and various alkalis and acids. The most common chemical disinfectants are the oxidizing chemicals, and of these, chlorine is the most widely used.[21]

Disinfectants act through one or more of a number of mechanisms, including damaging the cell wall, altering cell permeability, altering the colloidal nature of the protoplasm and inhibiting enzyme activity. In applying disinfecting agents, several factors need to be considered: contact time, concentration and type of chemical agent, intensity and nature of physical agent, temperature, number of organisms, and nature of suspending liquid.[27]

2.3.3.3. Dechlorination

Dechlorination is the removal of free and total combined chlorine residue from chlorinated wastewater effluent before its reuse or discharge to receiving waters. Chlorine compounds react with many organic compounds in the effluent to produce undesired toxic compounds that cause long-term adverse impacts on the water environment and potentially toxic effects on aquatic microorganisms. Dechlorination may be brought about by the use of activated carbon, or by the addition of a reducing agent such as sulfur dioxide (SO_2), sodium sulfite (Na_2SO_3), or sodium metabisulfite ($Na_2S_2O_5$). It is important to note that dechlorination will not remove toxic by-products that have already been produced.[27]

2.3.3.4. Adsorption

Adsorption is the process of collecting soluble substances within a solution on a suitable interface. The term adsorption refers to a process wherein a material is concentrated at a solid surface from its liquid or gaseous surroundings. The history of carbon adsorption in the purification of water dates back to ancient times.[28] Adsorption on porous carbons was described as early as 1550 BC in an ancient Egyptian papyrus and later by Hippocrates and Pliny the Elder, mainly for medicinal purposes. However, on scientific records, the phenomenon of adsorption was observed by C.W. Scheele in 1773 for gases exposed to carbon.[29] However, it was Kayser who introduced for the first time in 1881 the term adsorption to differentiate surface accumulation from intermolecular penetration. He postulated that the basic feature of an adsorption process is surface accumulation of material. It is now customary to differentiate two types of adsorption. If the attraction between the solid surface and the adsorbed molecules is physical in nature, the adsorption is referred to as physical adsorption (physiosorption). Generally, in physical adsorption, the attractive forces between adsorbed molecules and the solid surface are van der Waals forces and they being weak in nature result in reversible adsorption. On the other hand, if the attraction forces are due to chemical bonding, the adsorption process is called chemisorption. In view of the higher strength of the bonding in chemisorption, it is difficult to remove chemisorbed species from the solid surface.[30]

2.3.3.5. Ion exchange

Ion exchange is the displacement of one ion by another. The displaced ion is originally a part of an insoluble material, and the displacing ion is originally in solution. At the completion of the process, the two ions are in reversed places: the displaced ion moves into solution and the displacing ion becomes a part of the insoluble material. Two types of ion exchange materials are used: the cation exchange material and the anion exchange material. The cation exchange material exchanges cations, while the anion exchange material exchanges anions. The insoluble part of the exchange material is called the host. If $\mathrm{R}^{-\mathrm{n}}$ represents the host part and $\mathrm{C}^{+\mathrm{m}}$ the exchangeable cation, the cation exchange material may be represented by $(\mathrm{R}^{-\mathrm{n}})_r\,(\mathrm{C}^{+\mathrm{m}})_{r\mathrm{n/m}}$, where r is the number of active sites in the insoluble material, rn/m is the number of charged exchangeable particles attached to the host material, $-n$ is the charge of the host, and $+$m is the charge of the exchangeable cation. On the other hand, if R^{+0} represents the host part of the anion exchange material and $\mathrm{A}^{-\mathrm{p}}$ its exchangeable anion, the exchange material may be represented by $(\mathrm{R}^{+0})_r\,(\mathrm{A}^{-\mathrm{p}})_{r\mathrm{o/p}}$, where the subscripts and superscripts are similarly defined as those for the cation exchange material. Letting be the displacing cation from solution, the cation exchange reaction is

$$(\mathrm{R}^{-\mathrm{n}})_r(\mathrm{C}^{+\mathrm{m}})_{r\mathrm{n/m}} + r\mathrm{n/q}\ \mathrm{C}_\mathrm{s}^{+\mathrm{q}} \leftrightarrow (\mathrm{R}^{-\mathrm{n}})_r(\mathrm{C}_\mathrm{s}^{+\mathrm{q}})_{r\mathrm{n/q}} + r\mathrm{n/m}\ \mathrm{C}^{+\mathrm{m}}.$$

In addition, letting be the displacing anion from solution, the anion exchange reaction may be represented by

$$(\mathrm{R}^{+0})_r(\mathrm{A}^{-\mathrm{p}})_{r\mathrm{o/p}} + r\mathrm{o/t}\ \mathrm{A}_\mathrm{s}^{-\mathrm{t}} \leftrightarrow (\mathrm{R}^{+0})_r(\mathrm{A}_\mathrm{s}^{-\mathrm{t}})_{r\mathrm{o/t}} + r\mathrm{o/p}\ \mathrm{A}^{-\mathrm{p}}.$$

As shown by the previous equations, ion exchange reactions are governed by equilibrium. For this reason, effluents from ion exchange processes never yield pure

Table 1. Displacement Series for Ion Exchange.

La^{3+}	SO_4^{-2}
Y^{3+}	CrO_4^{-2}
Ba^{2+}	NO_3^-
Pb^{2+}	AsO_4^{-3}
Sr^{2+}	PO_4^{-3}
Ca^{2+}	MoO_4^{-2}
Ni^{2+}	I^-
Cd^{2+}	Br
Cu^{2+}	Cl^-
Zn^{2+}	F^-
Mg^{2+}	OH^-
Ag^+	—
Cs^+	—
Rb^+	—
K^+	—
NH^{4+}	—
Na^+	—
Li^+	—
H^+	—

water. Table 1 shows the displacement series for ion exchange materials. When an ion species high in the table is in solution, it can displace ion species in the insoluble material below it in the table and, thus, be removed from solution. As noted in this table, to remove any cation in solution, the displaceable cation must be the proton H^+; and to remove any anion, the displaceable anion must be the hydroxyl ion OH^-.

Originally, natural and synthetic alumino silicates, called zeolites, were the only ones used as exchange materials. Presently, they have been largely replaced by synthetic resins. Synthetic resins are insoluble polymers to which are added, by certain chemical reactions, acidic and basic groups called functional groups. These groups are capable of performing reversible exchange reactions with ions in solution. The total number of these groups determines the exchange capacity of the exchange material, while the type of functional group determines ion selectivity.

Ion exchange[31] shares various common features along with adsorption, in regard to application in batch and fixed-bed processes and they can be grouped together as "sorption processes" for a unified treatment to have high water quality.

2.3.3.5.1. *Adsorbent materials*

One of the most important characteristics of the adsorbent materials is the quantity of adsorbate it can accumulate which is usually calculated from the adsorption isotherms. The adsorption isotherms are constant-temperature equilibrium relationship between the quantity of adsorbate per unit of adsorbent (q_e) and its equilibrium solution concentration (C_e). Several equations or models are available that describe this function like the Freundlich and the Langmuir equations. A good sorbent[32] should generally possess a porous structure (resulting in high surface area) and the time taken for adsorption equilibrium to be established should be as small as possible, so that it can be used to remove dye wastes in lesser time.

Adsorbent materials can be categorized according to its accessibility into natural or synthetic adsorbents. They can be inorganic (minerals) or organic (generally polymer) based. The inorganic sorbent materials such as activated carbon, alumina, clays, zeolites, and silica gel, on the other hand, the organic sorbent materials include resins, polymeric compounds, etc.

2.3.3.5.2. *Application of low-cost alternative adsorbents in wastewater treatment*

Natural materials or the wastes/by-products of industries or synthetically prepared materials, which cost less and can be used as such or after some minor treatment as adsorbents are generally called low-cost adsorbents (LCAs). A protocol based on the numerous studies for the development, utilization, and application of LCAs generally adopted by researchers. The LCAs as reported in literature improve the collection of low-cost and nonconventional adsorbents, while being environmental acceptable. These low-cost alternative adsorbents[33] may be classified in two ways:

(i) *On basis of their availability*, i.e., (a) natural materials such as wood, peat, coal, and lignite, (b) industrial/agricultural/domestic wastes or by-products such as slag, sludge, flyash, bagasse flyash, and red mud, and (c) synthesized products.
(ii) *Depending on their nature*, i.e., (a) inorganic and (b) organic.

(a) Natural materials: Natural materials generally used as LCAs are the one existing in nature and used as such or with minor treatment. Some of the materials used are:

- *Wood*: The adsorbent was studied without any pretreatment and was sieved into different size ranges prior to use. Authors[34,35] suggested that because of its low cost, the wood adsorbent does not need to be regenerated after use and may be disposed off by burning and the heat so evolved can be used for generating steam.
- *Eucalyptus bark*: The adsorption on bark was suggested because of its high tannin content, the polyhydroxy polyphenol groups of tannin are thought to be the active species in the adsorption process.[36]

- *Natural coal*: The coal was sulfonated and heated in a water bath prior to the study as adsorbent.[37] In addition, coal-based sorbents, namely charfines, lignite coal, and bituminous coal, were studied as adsorbent, due to the presence of acidic groups (carboxyl and hydroxyl) in their structure, which by its rule resulted in a chemisorption mechanism.[38]
- *Peat*: Peat is one of the natural materials widely available and studied as an alternative adsorbent for different pollutants by a number of researchers. It covers 3% of the world's land area, and the production of peat for energy use was 70 million m^3 in 2005 thereby, making it important material. The major constituents of peat are lignin, cellulose, fulvic, and humic acid and the carbon content is generally just over 50%. Peat may be used either without any pretreatment or it may be modified with some chemical pretreatment to improve its sorption properties and selectivity.[39,40]
- *Chitin and chitosan*: These are mechanically tough polysaccharides with chemical structures similar to cellulose, studied as adsorbents. Chitin is a fairly abundant natural biopolymer and is generally found in the exoskeletons of crabs and other arthropods and also in the cell wall of some fungi whereas chitosan or glucosamine is a deacetylated derivative of chitin and can be chemically prepared from it.[41] Both chitin and chitosan are being used as an attractive source of adsorbents, especially for metal removal. Nevertheless, they are versatile materials and have been used successfully for the removal of dyes. These materials can be used in different forms, from flake-types to gels, bead-types or fibers. The materials were studied with and without chemical treatment, either by demineralization followed by deproteination, or only by one of the two steps.[42]
- *Biomass*: Biomass of nonliving dried roots of water hyacinth without any pretreatment was used as asorbent. It has a high potential as a sorbent due to its physicochemical characteristics due to it has different functional groups such as carboxyl, amino, and phosphate.[43] The sorption processes using biomass have some limitations such as the sorption process is generally slow, pH dependent, and the effluent treatment by columns is difficult due to the clogging effect.[44,45]
- *Cotton*: Cotton is one of the most important and widely used fibers by humans. Cotton found naturally and consisting cellulose exhibits excellent physical and chemical properties in terms of stability, water absorbency, and heavy metals and dye removal ability.[46]
- *Clays*: Among natural materials, clays occupy a prominent position being low cost, available in abundance, and having good sorption properties. There are various types of clays such as ball clay, bentonite, commonclay, sepiolite, fire clay, fuller's earth (attapulgite and montmorillonite varieties), and kaolin. World production of bentonite was approximately 11.7 Mt and fuller's earth production was estimated

to be 5.61 Mt. Sales reported by producers for absorbent uses were 3.92 Mt in 2005 and fuller's earth (montmorillonite type) accounted for w68% of the clay used for absorbents, followed by bentonite and a small amount of kaolin. In recent years, there has been an increasing interest in utilizing clay minerals such as bentonite, kaolinite, diatomite, and fuller's earth as adsorbents to remove not only inorganic but also organic molecules. The adsorption efficiency of clays generally results from a net negative charge on the structure of minerals. This negative charge gives clay the capability to adsorb positively charged species. Clay along with other low-cost materials, such as bagasse pith and maize cob, has been used as adsorbent for wastewater treatment.[47] Clays can be modified too to improve their sorption capacity by giving some treatments such as replacing cations present in clay, heat treatment, and acid treatment.[48]

- *Zeolites*: They are the only existing crystalline materials with a well-defined pore structure in the microporous range. Zeolites used for the heavy metals and dye removal have some similarities to clays, as the adsorption properties of zeolites also result mainly from their ion-exchange capabilities. Nevertheless, the sorption mechanism of zeolite particles is complex owing to several reasons such as their porous structure and surface charges.[49]

(b) Industrial/agricultural/domestic wastes or by-products:

In addition to the above-discussed natural materials, a number of agricultural wastes/by-products have also been investigated as adsorbents for the removal of pollutants by a number of workers. Shells of almond, hazelnut, poplar, walnut sawdust, and ground hazelnut shells were investigated. In addition, bone, peat processed, soil, nut shells, waste coir pith, bagasse pith, eucalyptus bark, cotton sunflower stalks, wheat straw, and bagasse pith were used as LCAs.[17]

3. Reuse of Waste Cooking Oil in Biodiesel Production

Waste cooking oils are generated locally, whenever food is cooked or fried in oil. Restaurant waste oils and rendered animal fats are less expensive than food-grade oils. The amount of waste cooking oil generated in each country is huge and varies depending on the use of vegetable oil. As the annual consumption of vegetable oil in Egypt exceeds million MT/year (1.248.000 MT in 2005 according to the official statistics of the Ministry of Internal Trade in Egypt and USDA). Egyptians eat fava beans as a source of protein in their diet and fried falafel is the most popular restaurants. As in case of French fries, fried potatoes, and fish, an estimate of the potential amount of waste cooking oil from the collection in European Union (EU) is approximately 0.7–1.0 Mt per year. Consequently, millions of liters of oil used for frying food.

3.1. *Disposal of Waste Cooking Oil*

The methods used in disposal of large amounts of waste cooking oil by following two methods. First is by feeding these mixtures to animals; since 2002, the European Union (EU) has enforced a ban for this way to dispose waste oils because, during frying, many harmful compounds are formed and, if the waste cooking oil is used as an additive to feeding mixtures for domestic animals, it can result in the return of harmful compounds back into the food chain via animal meat.[50] Second, by disposal of waste cooking oil into sewage systems, this method may contaminate environmental water.

Waste cooking oils must be disposed of safely or used in a way that is not harmful to human beings currently, the expensive and large quantity of waste cooking oil from households and restaurants are collected to beneficial reuse by environmental friendly way in biodiesel production.

3.2. *Why Using Waste Cooking Oil in Biodiesel Production*

Waste cooking oil offers significant potential as an alternative low-cost biodiesel feed stock that could partly decrease the dependency on petroleum based fuel.

3.3. *Advantages of Using Biodiesel Instead of Petroleum Diesel*

Biodiesel has several distinct advantages compared with petrodiesel in addition to being fully competitive with petrodiesel in most technical aspects:

- Derivation from a renewable domestic resource, thus reducing dependence on and preserving petroleum.
- Biodegradability.
- Reduction of most exhaust emissions (with the exception of nitrogen oxides, NOx).
- Higher flash point, leading to safer handling and storage.
- Excellent lubricity, a fact that is steadily gaining importance with the advent of low sulfur petrodiesel fuels, which have greatly reduced lubricity. Adding biodiesel at low levels (1–2%) restores the lubricity. Some problems associated with biodiesel are its inherent higher price, which in many countries is offset by legislative and regulatory incentives or subsidies in the form of reduced excise taxes, slightly increased NO_x exhaust emissions (as mentioned above), stability when exposed to air (oxidative stability), and cold flow properties that are especially relevant in North America. The higher price can also be (partially) offset by the use of less expensive feedstocks, which has sparked interest in materials such as waste oils (e.g., used frying oils).[51]

3.4. *What is Biodiesel?*

The major components of vegetable oils and animal fats are triacylglycerols (TAG; often also called triglycerides). Chemically, TAGs are esters of fatty acids (FA) with glycerol. The TAG of vegetable oils and animal fats typically contain several different FA. Thus, different FA can be attached to one glycerol backbone. The different FA that are contained in the TAG comprise the FA profile (or FA composition) of the vegetable oil or animal fat. Because different FAs have different physical and chemical properties the FA profile is probably the most important parameter influencing the corresponding properties of a vegetable oil or animal fat.[51]

3.5. *The Biodiesel Production Methods*

Vegetable oils have their own limitations to use in place of diesel due to their high viscosity. The high viscosity results in poor atomization of the fuel when the oil is injected into the combustion chamber of a compression ignition (CI) engine. This can lead to extended ignition delay periods for initial combustion to commence and consequently a reduction in engine performance due to incomplete combustion of the injected fuel particles.[52,53]

The methods available for reducing the viscosity of vegetable oils are:

- direct use of vegetable oils,
- dilution of oils,
- microemulsification of oils,
- pyrolysis and catalytic cracking, and
- transesterification of oils and fats.

3.5.1. *Direct Use of Vegetable Oils*

The use of vegetable oils as an alternative renewable fuel competing with petroleum was proposed in the early 1980s. The advantages of vegetable oils as diesel fuel are as follows[54]:

- portability,
- ready availability,
- renewability,
- higher heat content (about 88% of D2 fuel),
- lower sulfur content,
- lower aromatic content, and
- biodegradability.

Full combustion of a fuel requires in existence the amount of stoichiometric oxygen. However, the amount of stoichiometric oxygen generally is not enough for full combustion because the fuel is not oxygenated. The structural oxygen content of fuel increases the combustion efficiency of the fuel due to increased mixing of oxygen with the fuel during combustion. For these reasons, the combustion efficiency and cetane number of vegetable oil are higher than those of diesel fuel, and the combustion efficiency of methanol and ethanol is higher than that of gasoline. The disadvantages of vegetable oils as diesel fuel are[55]:

- higher viscosity,
- lower volatility, and
- the reactivity of unsaturated hydrocarbon chains.

Problems appear only after the engine has been operating on vegetable oils for longer periods, especially with direct-injection engines. The problems include (a) coking and trumpet formation on the injectors to such an extent that fuel atomization does not occur properly or is even prevented as a result of plugged orifices, (b) carbon deposits, (c) oil ring sticking, and (d) thickening and gelling of the lubricating oil because of contamination by the vegetable oils.[56]

Among the renewable resources for the production of alternative fuels, triglycerides have attracted much attention as alternative diesel engine fuels.[57] However, the direct use of vegetable oils and/or oil blends is generally considered to be unsatisfactory and impractical for both direct injection and indirect type diesel engines because of their high viscosities and low volatilities, injector coking and trumpet formation on the injectors, higher level of carbon deposits, oil ring sticking, and thickening and gelling of the engine lubricant oil, acid composition (the reactivity of unsaturated hydrocarbon chains), and free FA content.[56,58–60] Consequently, different methods have been considered to reduce the viscosity of vegetable oils such as dilution, microemulsification, pyrolysis, catalytic cracking, and transesterification.

Vegetable oils can be used as fuels for diesel engines, but their viscosities are much higher than that of common diesel fuel and require modifications of the engines.[61] Vegetable oils could only replace a very small fraction of transport fuel. Different methods of reducing the high viscosity of vegetable oils have been considered:

1. Dilution of 25 parts of vegetable oil with 75 parts of diesel fuel.
2. Microemulsions with short-chain alcohols such as ethanol or methanol.
3. Pyrolysis and catalytic cracking, which produce alkanes, cycloalkanes, alkenes, and alkylbenzenes.
4. Transesterification with ethanol or methanol, which produces biodiesel.

3.5.2. *Dilution of Oils*

Dilution of oils with solvents and microemulsions of vegetable oils lowers the viscosity and mitigates some engine performance problems such as injector coking and carbon deposits, etc. To dilute vegetable oils, the addition of 4% ethanol to the oils increases the brake thermal efficiency, brake torque, and brake power while decreasing brake-specific fuel consumption. Since the boiling point of ethanol is less than that of vegetable oils, it could assist in the development of the combustion process through an unburned blend spray.[62] The viscosity of oil can be lowered by blending with pure ethanol. Twenty-five parts of sunflower oil and 75 parts of diesel were blended as diesel fuel.[63] The viscosity was 4.88 cSt at 313 K, while the maximum specified ASTM value was 4.0 cSt at 313 K. This mixture was not suitable for long-term use in a direct injection engine. Another study was conducted using the dilution technique on the same frying oil.[64] The addition of 4% ethanol to D2 fuel increases the brake thermal efficiency, brake torque, and brake power while decreasing brake-specific fuel consumption. Since the boiling point of ethanol is less than that of D2 fuel, it could assist the development of the combustion process through an unburned blend spray.[62]

3.5.3. *Microemulsion of Oils*

Defined as a colloidal equilibrium dispersion of optically isotropic fluid microstructure with dimensions generally in the 1–150 nm range formed spontaneously from two immiscible liquids and one or more ionic or nonionic amphiphiles.[65,66] The short engine performances of both ionic and nonionic microemulsions of ethanol in soybean oil were nearly as good as that of D2 fuel.[67] To solve the problem of the high viscosity of vegetable oils, microemulsions with solvents such as methanol, ethanol, and 1-butanol have been studied. All microemulsions with butanol, hexanol, and octanol met the maximum viscosity requirement for D2 fuel. The 2-octanol was found to be an effective amphiphile in the micellar solubilization of methanol in triolein and soybean oil.[65,68] Ziejewski *et al.*[69] prepared an emulsion of 53% (vol) alkali-refined and winterized sunflower oil, 13.3% (vol) 190 proof ethanol, and 33.4% (vol) 1-butanol. This nonionic emulsion had a viscosity of 6.31 cSt at 313 K, a cetane number of 25, and an ash content of less than 0.01%. Lower viscosities and better spray patterns (more even) were observed with an increase of 1-butanol. In a 200-h laboratory screening endurance test, no significant deteriorations in performance were observed, but irregular injector needle sticking, heavy carbon deposits, incomplete combustion, and an increase in lubricating oil viscosity were reported.[65] A microemulsion prepared by blending soybean oil, methanol, 2-octanol, and cetane improver in the ratio of 52.7:13.3:33.3:1.0 also passed the 200-h EMA test.[70] Schwab *et al.*[68] used the ternary phase equilibrium diagram and the plot of viscosity versus solvent fraction to determine the emulsified fuel formulations. All microemulsions with butanol, hexanol, and octanol met the maximum viscosity requirement for D2. The 2-octanol was an effective amphiphile in the micellar solubilization of methanol in triolein and soybean oil. Methanol was often used due to economic advantage over ethanol. This method produce lower fuel viscosities, better spray patters during combustion but the main disadvantage of this method is to have lower energy content and cetane number.

3.5.4. *Pyrolysis and Catalytic Cracking*

Pyrolysis is the conversion of one substance with long chain and saturated substance into another by means of heat or by heat with the aid of a catalyst.[71] It involves heating in the absence of air or oxygen and cleavage of chemical bonds to yield small molecules.[72] The pyrolyzed material can be vegetable oils, animal fats, natural FAs, and methyl esters of FAs. The pyrolysis of fats has been investigated for more than 100 years, especially in those areas of the world that lack deposits of petroleum. The first pyrolysis of vegetable oil was conducted in an attempt to synthesize petroleum from vegetable oil. The mechanisms for the thermal decomposition of a triacylglyceride are given in Fig. 2.

Soybean oil was thermally decomposed and distilled in air and nitrogen sparged with a standard ASTM distillation apparatus.[73,74] The main components were alkanes and alkenes, which accounted for *ca.* 60% of the total weight. Carboxylic acids accounted for another 9.6–16.1%. Catalytic cracking of vegetable oils to produce biofuels has been studied by Pioch *et al.*[75] Copra oil and palm oil stearin were cracked over a standard petroleum catalyst SiO_2/Al_2O_3 at 723 K to produce gases, liquids, and solids with lower molecular weights. The condensed organic phase was fractionated to produce biogasoline and biodiesel fuels. However, the disadvantage of this process is energy intensive and hence higher cost.

3.5.5. *Transesterification of Oils and Fats*

Among all these alternatives, transesterification seems to be the best choice, as the physical characteristics of FA esters (biodiesel) are very close to those of diesel fuel

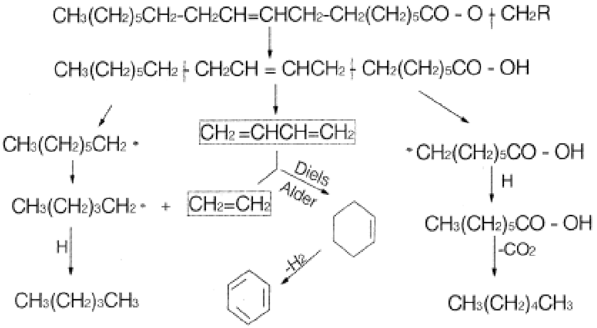


Figure 2. The Mechanism of Thermal Decomposition of Triglycerides.

and the process is relatively simple as (renew ability, higher cetane number, lower emissions, higher consumption efficiency.[66,76,77] In the esterification of an acid, an alcohol acts as a nucleophilic reagent; in the hydrolysis of an ester, an alcohol is displaced by a nucleophilic reagent. This alcoholysis (cleavage by an alcohol) of an ester is called transesterification.[78] Transesterified vegetable oils have proven to be a viable alternative diesel engine fuel with characteristics similar to those of diesel fuel. The transesterification reaction proceeds with catalyst or any unused catalyst by using primary or secondary monohydric aliphatic alcohols having between one and eight carbon atoms as follows:

$$\text{Triglycerides} + \text{monohydric alcohol} \longleftrightarrow \text{glycerine} + \text{monoalkyl esters}$$

Transesterification is catalyzed by a base (as NaOH, KOH, carbonates, and corresponding sodium and potassium alkoxides such as sodium methoxide, sodium ethoxide, sodium propoxide, and sodium butoxide) or acid (H_2SO_4 or dry HCl) or enzymes (as lipases). Alkali-catalyzed transesterification is much faster than acid-catalyzed transesterification and is most often used commercially. The transesterification is an equilibrium reaction. To shift the equilibrium to the right, it is necessary to use a large excess of the alcohol or else to remove one of the products from the reaction mixture. Furthermore, the methyl or ethyl esters of FAs can be burned directly in unmodified diesel engines, with very low deposit formation. Although short-term tests using neat vegetable oil showed promising results, longer tests led to injector coking, more engine deposits, ring sticking, and thickening of the engine lubricant. These experiences led to the use of modified vegetable oil as a fuel. Technical aspects of biodiesel are close to petroleum diesel, such as physical and chemical characteristics of methyl esters related to its performance in CI engines.[79] Compared with transesterification, pyrolysis has more advantages. The liquid fuel produced from pyrolysis has similar chemical components to conventional petroleum diesel fuel.[80] However, these methods have a problem in disposal of by product (glycerol and wastewater).[66,76,77]

3.6. *Catalyst Types*

Transesterification is the process of employing an alcohol (usually methanol, ethanol, or butanol) in the presence of a catalyst to obtain methyl or ethyl or butyl esters of the oil with glycerol as a by product. Transesterification can be carried out catalytically in different routes namely: (i) base, (ii) acid, and (iii) enzyme catalyst; Table 2 summarize the advantages and disadvantages for the different types used in the biodiesel production.

Table 2. Advantages and Disadvantages at Different Types of Catalysts used in the Biodiesel Production.

Type	Example	Advantages	Disadvantages	References
Alkali				
• Homogenous	NaOH, KOH	High catalytic activity, low cost, favorable kinetics, modest operation conditions	Low FFA requirement, anhydrous conditions, saponification, emulsion formation, more wastewater from purification, disposable	[81,82–85]
• Heterogeneous	CaO, $CaTiO_3$, $CaZrO_3$, CaO-CeO_2, $CaMnO_3$, $CaFeO_5$, KOH/Al_2O_3, KOH/NaY, Al_2O_3/KI, ETS-10 zeolite, Alumina/silica supported K_2CO_3	Noncorrosive, environmentally benign, recyclable, fewer disposal problems, easily separation, higher selectivity, longer catalyst lifetimes	Low FFA requirement, anhydrous conditions, more wastewater from purification, high molar ratio of alcohol to oil requirement, high reaction temperature and pressure, diffusion limitation, high cost	[82, 84–88]
Acid				
• Homogenous	Concentrated sulfuric acid	Catalyze esterification and transesterification simultaneously, avoid soap formation	Equipment corrosion, more waste from neutralization, difficult to recycle, higher reaction temperature, long reaction times, weak catalytic activity	[81–83]
• Heterogeneous	ZnO/I_2, ZrO_2/SO_4^{2-}, TiO_2/SO_4^{2-}, carbon-based solid acid catalyst, carbohydrate-derived catalyst, vanadyl phosphate, niobic acid, sulphated zirconia, Amberlyst-15, Nafion NR50	Catalyze esterification and transesterification simultaneously, recyclable, eco-friendly	Low acid site concentrations, low microporoisty, diffusion limitations, high cost	[82,85,89,90]
Enzymes	*Candida antarctica* fraction B lipase, *Rhizomucor mieber* lipase	Avoid soap formation, nonpolluting, easier purification	Expensive, denaturation	[85,89,91]

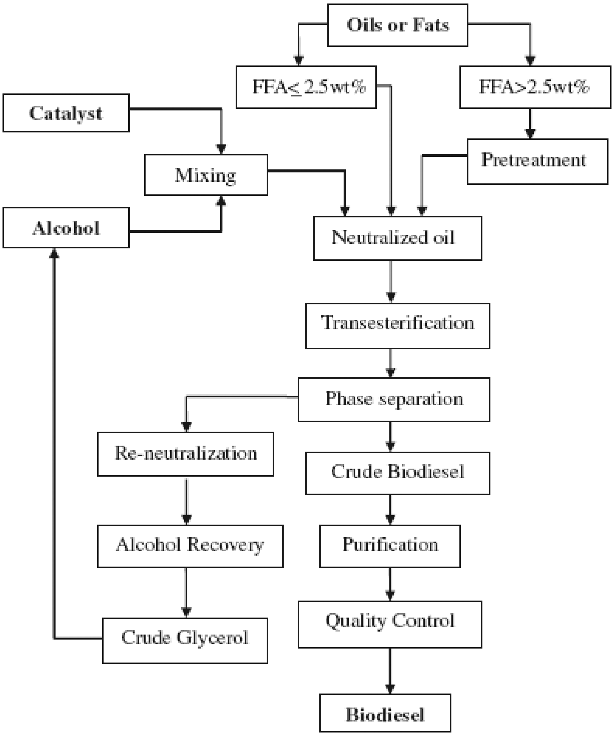


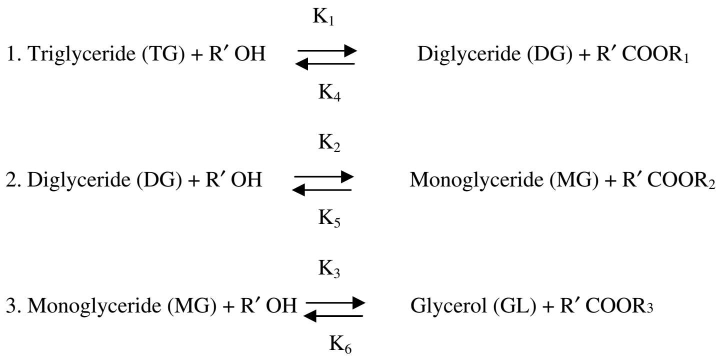
Figure 3. Simplified Process Flowchart of Alkali-Catalyzed Biodiesel Production.

3.7. *Biodiesel Production Process*

Most of the biodiesel is produced by the alkali-catalyzed process. Figure 3 shows a simplified flowchart of the alkali-catalyst process. As feedstocks with high free FA will react undesirably with the alkali catalyst thereby forming soap. The maximum amount of free FAs acceptable in an alkali-catalyzed system is below 2.5 wt.% FFA. If the oil or fat feedstock has an FFA content over 2.5 wt.%, a pretreatment step is necessary before the transesterification process.[92]

3.8. *The Mechanism and Kinetics*

Transesterification consists of a number of consecutive and reversible reactions. The triglyceride is converted stepwise to diglyceride, monoglyceride, and finally glycerol (Fig. 4). A mole of ester is liberated at each step. The reactions are reversible, although the equilibrium lies toward the production of FA esters and glycerol. The reaction mechanism for alkali-catalyzed transesterification was formulated as three steps. The first step is an attack on the carbonyl carbon atom of the triglyceride molecule by the anion of the alcohol (methoxide ion) to form a tetrahedral intermediate. In

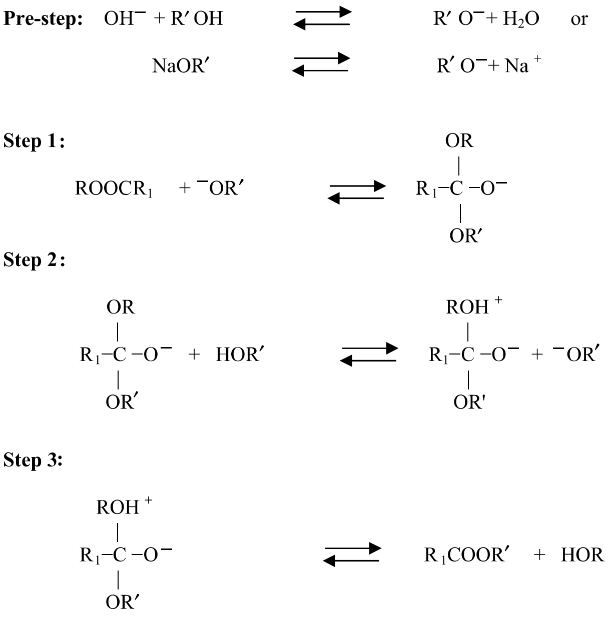


$$\text{1. Triglyceride (TG)} + R'OH \underset{K_4}{\overset{K_1}{\rightleftarrows}} \text{Diglyceride (DG)} + R'COOR_1$$

$$\text{2. Diglyceride (DG)} + R'OH \underset{K_5}{\overset{K_2}{\rightleftarrows}} \text{Monoglyceride (MG)} + R'COOR_2$$

$$\text{3. Monoglyceride (MG)} + R'OH \underset{K_6}{\overset{K_3}{\rightleftarrows}} \text{Glycerol (GL)} + R'COOR_3$$

Figure 4. The Transesterification Reactions of Vegetable Oil with Alcohol to Esters and Glycerol.



Pre-step:

$$OH^- + R'OH \rightleftarrows R'O^- + H_2O \quad \text{or}$$

$$NaOR' \rightleftarrows R'O^- + Na^+$$

Step 1:

$$ROOCR_1 + {}^-OR' \rightleftarrows R_1{-}\overset{\displaystyle OR}{\underset{\displaystyle OR'}{C}}{-}O^-$$

Step 2:

$$R_1{-}\overset{\displaystyle OR}{\underset{\displaystyle OR'}{C}}{-}O^- + HOR' \rightleftarrows R_1{-}\overset{\displaystyle ROH^+}{\underset{\displaystyle OR'}{C}}{-}O^- + {}^-OR'$$

Step 3:

$$R_1{-}\overset{\displaystyle ROH^+}{\underset{\displaystyle OR'}{C}}{-}O^- \rightleftarrows R_1COOR' + HOR$$

Figure 5. The Mechanism of Alkali-Catalyzed Transesterification of Triglycerides with Alcohol.

the second step, the tetrahedral intermediate reacts with an alcohol (methanol) to regenerate the anion of the alcohol (methoxide ion). In the last step, rearrangement of the tetrahedral intermediate results in the formation of an FA ester and a diglyceride. When NaOH, KOH, K_2CO_3, or other similar catalysts were mixed with alcohol, the actual catalyst, alkoxide group is formed. A small amount of water, generated in the reaction, may cause soap formation during transesterification. Figure 5 summarizes the mechanism of alkali-catalyzed transesterification.[93]

3.9. *Main Factors Affecting the Yield of Biodiesel*

3.9.1. *Alcohol Quantity*

Many researchers recognized that one of the main factors affecting the yield of biodiesel is the molar ratio of alcohol to triglyceride.[65,94–96] Theoretically, the ratio for transesterification reaction requires 3 mol of alcohol for 1 mol of triglyceride to produce 3 mol of FA ester and 1 mol of glycerol. An excess of alcohol is used in biodiesel production to ensure that the oils or fats will be completely converted to esters and a higher alcohol–triglyceride ratio can result in a greater ester conversion in a shorter time. The yield of biodiesel is increased when the alcohol–triglyceride ratio is raised beyond 3 and reaches a maximum. Further increasing the alcohol amount beyond the optimal ratio will not increase the yield, but will increase cost for alcohol recovery.[94] In addition, the molar ratio is associated with the type of catalyst used and the molar ratio of alcohol to triglycerides in most investigations is 6:1, with the use of an alkali catalyst.[95,96] When the percentage of FA acids in the oils or fats is high, such as in the case of waste cooking oil, a molar ratio as high as 15:1 is needed when using acid-catalyzed transesterification.[94,97–99]

3.9.2. *Reaction Time*

Freedman *et al.*[100] found that the conversion rate of FA esters increases with reaction time. At the beginning, the reaction is slow due to the mixing and dispersion of alcohol into the oil. After a while, the reaction proceeds very fast. Normally, the yield reaches a maximum at a reaction time of <90 min, and then remains relatively constant with a further increase in the reaction time.[94,100] Moreover, excess reaction time will lead to a reduction in the product yield due to the backward reaction of transesterification, resulting in a loss of esters as well as causing more FAs to form soaps.[101,102]

3.9.3. *Reaction Temperature*

Temperature clearly influences the reaction and yield of the biodiesel product. A higher reaction temperature can decrease the viscosities of oils and result in an increased reaction rate, and a shortened reaction time. However, Leung[94] and Eevera *et al.*[101] found that when the reaction temperature increases beyond the optimal level, the yield of the biodiesel product decreases because a higher reaction temperature accelerates the saponification reaction of triglycerides. The reaction temperature must be less than the boiling point of alcohol in order to ensure that the alcohol will not leak out through vaporization. Depending on the oil used, the optimal temperature ranges from 50°C to 60°C.[65,94,100]

3.9.4. *Catalyst Concentration*

Catalyst concentration can affect the yield of the biodiesel product. As mentioned before, the most commonly used catalyst for the reaction is sodium hydroxide. However, Freedman *et al.*[100] found that sodium methoxide was more effective than sodium hydroxide because upon mixing sodium hydroxide with methanol a small amount of water will be produced, which will affect the product yield because of the hydrolysis reaction.[103] This is the reason why the catalyst should be added into the methanol first and then mixed with the oil. As the catalyst concentration increases, the conversion of triglyceride and the yield of biodiesel increase. This is because an insufficient amount of catalysts result in an incomplete conversion of the triglycerides into the FA esters.[94,103] Usually, the yield reaches an optimal value when the catalyst (NaOH) concentration reaches 1.5 wt.% and then decreases a little with a further increase in catalyst concentration. The reduction of the yield of the biodiesel is due to the addition of excessive alkali catalyst causing more triglycerides to react with the alkali catalyst and form more soap.[94,101]

3.10. *Analytical Methods for Biodiesel*

There are a variety of standard methods for analyzing biodiesel content in mixtures, including gas chromatography (GC), high-performance liquid chromatography (HPLC), gel permeation chromatography (GPC), proton nuclear magnetic resonance spectroscopy (1H NMR), near-infrared (NIR) spectroscopy, TLC, and Fourier transform infrared spectroscopy (FTIR). In the literature, the transesterification reactions were monitored by GC or TLC. Determination of products by GC is not simple, but time consuming and difficult because of the need for sample derivatization or the use of special columns. TLC can be simple but it gives a number of qualitative results. Moreover, it can be a problem to differentiate high molecular weight species such as monoglycerides and diglycerides. Instead of this techniques, thermogravimetric analysis (TGA) is used. TGA is a technique used for characterizing thermal stability of a material (compound or mixture) by measuring changes in its physicochemical properties expressed as weight change as a function of increasing temperature. Therefore, the change in mass of a substance is measured as a function of increasing temperature and the change is correlated to the thermal stability of a material, which is directly related to the material's volatility or thermal degradation to gaseous products. It is proposed that TGA can be used as a fast and low-cost screening method of biodiesel conversion. Thus, the biodiesel industry may be able to use this technique to monitor the yield of biodiesel in order to aid in determining the complete conversion time. It is important to note that it is proposed that this technique is used only to monitor transesterification progress and not to determine biodiesel product quality. This can ensure that the conversions discontinued at the proper time will lead to reduced production time, thus increasing the overall efficiency of the biodiesel industry. The TGA technique has been used to characterize biodiesel and oils,[19–21] but these studies did not specifically evaluate the use of TGA as a method for evaluating its efficacy for measuring the extent of oil to biodiesel conversion at any time during the transesterification process, as proposed in the present work. The TGA instrument consists of a precision mass balance, which records the initial and instantaneous mass of a sample, and a furnace, which increases the temperature in a linear relationship with time (the range of temperatures obtained typically being between 20°C and 1000°C). Mass measurements are performed in air or in an inert gas such as nitrogen or argon. The advantages and limitations of TGA as a method of quantitating biodiesel production in comparison to other methods are tabulated in Table 3.[104]

4. Plastic Waste Recycling

4.1. *History of Plastics Production*

From a historical viewpoint, the development of plastics can be regarded as one of the most important technical achievements of the 20th century. In just 50 years, plastics have permeated virtually every aspect of daily life, paving the way for new inventions, and replacing materials in existing products. Plastics are light, durable, and versatile and they are now used in countless applications. The success of these materials has

Table 3. Comparison of TGA with other Methods Known for the Quantitating of Biodiesel.

Quantification method	Advantages	Disadvantages
TGA	No reagent or solvent required Differentiates easily biodiesel and plant oil Relatively inexpensive	Does not differentiate among different FA methyl esters in biodiesel
^{1}HNMR	Simple, accurate, and precise in determining the biodiesel content	Deuterated solvents required ($CDCl_3$ in the present case)
GC	Can differentiate among glycerides, methyl esters, and glycerol	Instrumentation relatively more expensive Standard solutions are required
NIR	No solvent required Can differentiate among soybean oil, biodiesel, and glycerol	Cumbersome to calculate direct conversion Cannot qualify low levels of contaminants
HPLC	Differentiates among mono-, di-, and triglycerides, methyl esters and glycerol	External solvents and standards are required to determine particular components of the mixture Cumbersome to calculate direct conversion

been based on their properties of resilience, resistance to moisture, chemicals, and photo- and biodegradations, their stability, and the fact that they can be molded into any desired form. The plastics industry is relatively young. The first plastics were produced at the turn of the century, and were based mainly on natural raw materials. Only in 1930, thermoplastics, made from the basic materials styrene, vinyl chloride, and ethylene, were introduced onto the market. However, the main growth of the plastics industry did not take place before the 1960s, reaching a peak in 1973, when production reached over 40 million tonnes per year. Following a temporary drop in production during the oil crisis and the economic recession in the beginning of the 1980s, the world production of plastics continued to increase, to approximately 77 million tonnes in 1986 and 86 million tonnes in 1990. Figure 6 shows the rapid development of plastics production worldwide, which now far exceeds the combined production of nonferrous metals such as aluminium, zinc, lead, and copper. The major producers and consumers of plastic materials are North America, Western Europe, and Japan. Currently, about three-quarters of the world's plastics are produced in these regions. Table 4 gives an indication of the shares of plastics production in the main world regions in the late 1980s. It can be seen that the share of "other" countries, which include the whole of Africa, is small, even compared with countries in Central and South America and in Asia. However, considerable changes are taking place. Western Europe's share of production is in decline, and capacity plans indicate that the shares of Asia (excluding Japan), Central and South America, and the Middle East will increase rapidly at the expense of the traditional producers.

In contrast with most African countries, which import plastics, Egypt has rapidly increased its output of plastics, largely because of the government's policy to encourage domestic production. The total value of the plastics produced in Egypt increased from \$18.5 million in 1979 to more than \$200 million in 1991. A part of this output is exported, and total export earnings rose from about \$0.5 million in the early 1980s, to more than \$60 million in 1990.[105]

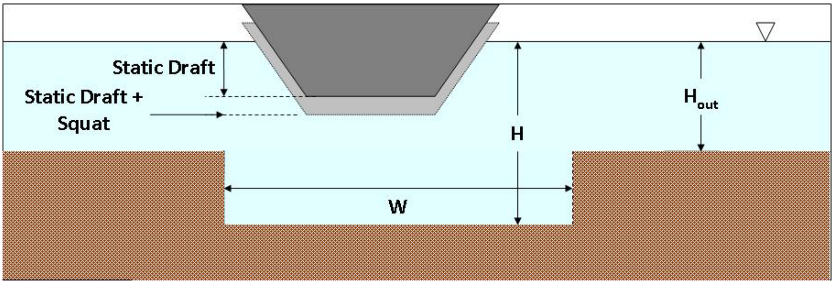


Figure 6. Development of Plastics Production Worldwide.

Table 4. Location of World Plastics Production in the late 1980s (%).

North America	32.0
Western Europe	31.2
Eastern Europe and the former Soviet Union	12.3
Japan	11.7
Other Asian countries (excluding Japan)	6.9
Central and South America	4.3
Other	1.6

The growth of plastics production has also resulted in the generation of a vast waste stream that needs to be properly managed to avoid environmental damage.

4.2. *Plastic Waste*

Consumption of plastic products has increased dramatically over the past few decades. Accordingly, the amounts of waste plastics have also increased. For example, the annual plastic waste stream generated in Western Europe is estimated at around 21.5 million tons. Over 78 wt% of the plastic waste corresponds to thermoplastics (primarily LDPE and high-density polyethylene (HDPE) (LDPE-HDPE), PP, PS, PVC, and polyethylene terephthalate (PET)). The remaining 22 wt% corresponds to thermosets (mainly epoxy resins and polyurethane).[106] The majority of plastic waste comes from the post-user market. Post-user wastes are found mainly in MSW, as well as in the following economic sectors: agriculture, commercial centers, C & D, and automotive, electrical, and electronics industries. Pre-user plastic wastes, which generally account for less than 10%, are generated during the manufacture of virgin plastic from raw materials (oil, natural gas, salt, etc.) and from the conversion of plastics into plastic products.

Owing to the increasing amounts of plastic waste, the municipal collection and disposal services are woefully insufficient, particularly in low-income areas, where waste accumulates in the streets and poses risks to human health. Therefore, the application of novel recycling technologies capable of absorbing large amounts of plastic waste is necessary to protect the environment and public health.[107]

4.3. *Plastic Waste Management*

Figure 7 provides a general view of the alternative plastic waste management methods. Despite recent advances, 61 wt% of all the waste plastics generated all over the world are still disposed of to landfill, a management alternative that generates serious environmental problems due to their low density, resistance to biological degradation, and combustible nature. Incineration with energy recovery represents the main alternative to landfill, absorbing on average 23 wt% of the plastic waste.

Energy recovery involves the use of waste plastics as fuels either in purposely built incineration facilities for the production of heat or electricity, or in specially adapted energy intensive industrial processes such as cement kilns and boilers for steam or heat production. Technical advances in the design and operation of incineration facilities and the application of strict regulatory measures means that energy recovery is usually regarded as an environmentally sound option.

Recycling takes the remaining 16 wt%. Recycling processes for waste plastics are classified into two categories: mechanical and feedstock recyclings. The former involves reprocessing of thermoplastic waste into secondary plastic products. The processes employed in the waste plastics reprocessing industries are basically the same as those used in manufacturing products from virgin plastics. Small factories based on this technology are common all over the world for the recycling of polyethylene or pp. However, owing to incomplete separation of different plastic types, the presence of additives and contaminants, and also due to the modification of the original polymeric structure during its first use (typically chain cross-linking reactions), mechanically recycled plastics only find use in lower grade applications. The latter, which currently absorbs only 1.6 wt% of the plastic waste, includes a variety of processes such as pyrolysis, catalytic conversion, depolymerization

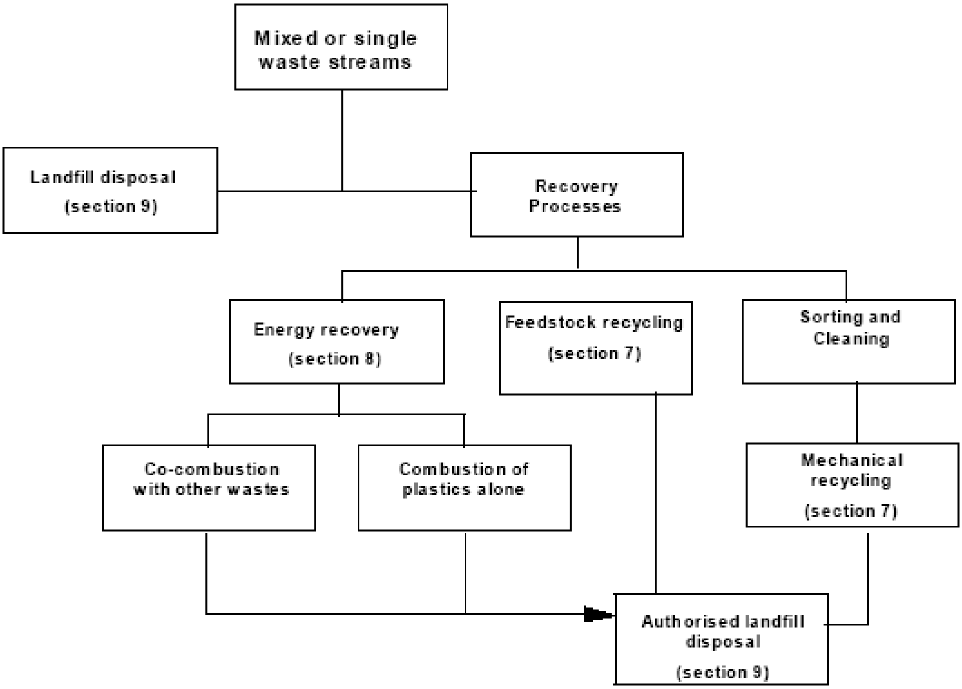


Figure 7. Integrated Approach to Plastic Waste Management.

and gasification, designed to transform plastic waste into hydrocarbon products for use in the preparation of virgin polymers, refined chemicals, or fuels. Despite not being used extensively, feedstock recycling is attracting increasing scientific and commercial attention as an alternative with the potential to absorb very large amounts of plastic wastes. The application of catalysts is giving feedstock recycling a new impulse that is expected to put this technology at the forefront of plastic management.

The shortage of suitable land for waste disposal sites, strict environmental legislation and controls, and the increasing amounts of hazardous waste have all contributed to a rapid increase in the cost of disposal services. Consequently, policy has shifted away from the acceptance of simply throwing away plastic waste, toward the encouragement of both the prevention of waste generation and the minimization of the amounts of waste that need to be disposed of, including the elimination of impediments and distortions that encourage the overproduction of wastes, valorization by energy recovery, recycling, and residual management.[106,107]

Mechanical recycling represents the second worldwide largest recovery option. Improvements in the sorting and cleaning of plastic waste are expected to facilitate an even wider the application of this technique. This section provides a general view of the worldwide mechanical recycling activities, paying special attention to the current stages involved in mechanical recycling.

4.4. *Mechanical Recycling*

Mechanical recycling in developing countries is often quite different from that in the industrialized countries of Western Europe and North America, largely because of the very different economic circumstances.[108]

4.4.1. *Mechanical Recycling in Industrialized Countries*

In the industrialized countries, a distinction between the reprocessing of primary and secondary wastes is relevant. Primary waste refers to the pre-user plastic waste. It is pure and its quality is almost as high as that of virgin plastics. Therefore, it is suitable for reprocessing with standard equipment into the same kind of products manufactured from virgin plastics. Reprocessing of primary waste into products with characteristics similar to those of the original products is called primary recycling.

Plastic goods cannot be produced without generating waste, so that primary recycling has been practiced since the establishment of the industry. As a result, it has been subjected to extensive scientific research, for example, on the melting behavior and thermal stability of plastics during the remelting process. Nowadays, almost all major plastics manufacturers are engaged in some form of plastics reprocessing.

The term “secondary waste” refers to the post-user plastic waste. These waste plastics are impure, i.e., they may be contaminated and often consist of mixtures of various types of plastics. The direct reprocessing of such mixed supplies (so-called secondary recycling) results in “alloys” with poor mechanical properties, because of the different characteristics of the plastics they contain. The potential for marketing these materials is relatively low. Thus, the secondary waste should be sorted and cleaned, before reprocessing for improving the quality of the final products manufactured from it.

In the early 1980s, there was a noticeable increase in interest and activities in waste plastics reprocessing. For mixed and contaminated waste plastics, both manufacturing processes and markets have been developed to produce bulky final products. Product applications include fence posts and garden furniture, where the plastics are used to replace wood, for which relatively high levels of impurities can be tolerated. At first, it was rather difficult to market such products, but recently the enduses of reprocessed mixed plastics has expanded considerably. However, most factories involved in the production of such secondary products still use relatively pure and well-defined waste plastics from industrial or commercial sources, since such supplies are available and guarantee a better quality of the end products. In general, reprocessing is not a problem if the supply of waste plastics is pure (i.e., homogeneous and uncontaminated), and is obtained from industrial or commercial sources such as workshops or supermarkets, agriculture, etc. Problems arise when the waste to be reprocessed is highly mixed and contaminated, such as household waste. In such case, the identification of the types of constituent plastics is essential but difficult step in the plastics recovery process.

Although technologies have been developed to separate and clean mixed and contaminated supplies, they do not as yet operate successfully on a commercial basis. The opportunities for recovery are greater if the waste materials are separated before they are collected for recycling. Clear labeling of the various types of plastics during manufacturing would facilitate separation at a later stage, and would increase the efficiency of the recycling process.

In the industrialized countries, both primary and secondary recyclings are based on capital-intensive, labor-extensive processing equipment, and mass production in order to ensure the maximum return on investments. Both the technical and economic feasibilities of the technology for the recovery of mixed plastics remain to be proven. For these and other reasons, the recycling of plastic waste has taken much longer to become established than that of other waste materials such as paper and glass. Despite the progress made so far, plastics recovery still has a long way to go. The gap between the amount of waste plastics generated and that recovered is still enormous.[108]

4.4.2. *Mechanical Recycling in Developing Countries*

In the developing countries, the distinction between primary and secondary wastes is not as pronounced as in the industrialized countries. Although the plastics manufacturing industries themselves recover most of their primary waste, the general picture is one in which all kinds of waste plastics are recovered (primary and secondary waste plastics). In contrast with the situation in industrialized countries, no markets have been developed for new product applications based on secondary waste plastics. Instead, the same types of regularly available products are manufactured from secondary waste plastics, although they are sold at lower prices and are poorer in quality. The reprocessing sector in the developing countries can be characterized as follows:

- The markets for low-cost consumer items are extensive. Because of the large numbers of low-income consumers, the level of market acceptance of cheaper and poorer-quality products is high. Within low-income groups, the demand for such products is generally greater than that for products made from more expensive virgin plastics.
- High unemployment and low labor costs mean that the labor-intensive manual processes involved in the reprocessing of waste plastics, such as collecting, washing, and sorting waste, are economically feasible.
- There are few or no regulations or quality standards for recycled products.

The plastic materials recovered from waste are always inferior to the originals. In the industrialized countries, such inferior quality products may not pass standard quality tests, but in the developing countries, they are tolerated.

Large plastics factories usually do not reprocess municipal plastic waste because of the sensitivity of their machinery to the impurities that are normally present in such waste, the irregular shapes of plastic pellets, and the generally low quality of the final products made from waste materials. However, large factories sometimes recycle their own plastic waste, thus ensuring a hygienic production process, i.e., recovered materials are usually mixed with varying proportions of virgin raw materials.

In contrast with the industrialized countries, most plastics are reprocessed in (informal) small-scale enterprises that depend almost exclusively on plastic waste. Small and low-technology workshops generally produce goods for the low-cost and low-quality market, and thus attempt to minimize costs by saving on raw materials. The technologies they use are in principle the same as those used in the (formal) large industries, although most machinery is outdated or has been upgraded with locally available spare parts. Given their small size and the large proportion of scrap they use, these enterprises are highly dependent on a network of dealers and reprocessors for supplies of raw materials that meet their specifications.[108]

4.4.3. *Mechanical Recycling Stages*

For improving considerably the quality of the final products manufactured from secondary plastic waste, the collected secondary waste should initially be cleaned and sorted before size reduction. Then, it can be reprocessed into various plastic products.[109]

4.4.3.1. Collection

Waste plastics from municipal sources are collected by hand and are roughly preselected by waste pickers or primary traders. This stage is labor-intensive and requires little or no capital investment. There are several points within an MSW system where waste can be retrieved for recovery: at source, i.e., directly from private homes; from waste bins; from refuse collection vehicles; and at municipal waste dumps.

4.4.3.2. Cleaning

The cleaning stage consists of washing and drying the waste plastic items.

4.4.3.2.1. *Washing*

Waste plastics must be washed for removing most contaminants (nonplastics, dirt, etc.) prior to reprocessing. Clean waste materials get better prices and their final products have improved quality. Films and rigid waste plastics are usually cleaned before the size reduction stage. Foreign materials such as glued paper labels are also removed. Rigid plastics are often washed a second time after they are shredded. The material can be washed manually or mechanically. If the waste is greasy, hot water with soap, detergent, or caustic soda should be used.

4.4.3.2.2. *Drying*

After washing, waste plastics can be dried either manually or mechanically. With the manual method, the plastics are spread out in the sun to dry. Plastic films can be hung on lines, and thus require only half the area normally used when plastics are spread out to dry. After they have been shredded, waste plastics may be either centrifuged or spread out in layers several centimetres thick to dry. The fan blows heated air underneath the shredded plastic, thus reducing its moisture content.

4.4.3.3. Sorting

The degree of sorting of plastic waste varies considerably, depending on the demand and the special wishes of the manufacturers to whom it will be sold. The material may be sorted according to color, type of plastic, etc.[110]

4.4.3.3.1. *Identifying types of plastics*

Waste plastics must be correctly identified. If not, this can create severe problems during reprocessing, leading to products with a poor appearance and impaired mechanical properties.

It is usually difficult to identify exactly which type of plastic is present solely from the type of product. Many different types of plastics may look identical or one type of plastic may appear to have several physical and chemical characteristics depending on the type of additive that has been used. Detailed chemical tests may be needed to make a definite identification of a polymer.[109] However, experience in this field can be gained with practice, and in case of doubt, testing is the only option. Some simple tests using basic equipment can provide adequate information for identification. However, when in doubt, multiple tests could be applied to distinguish the various polymers:

- To make a general distinction between thermoplastics and thermosets, take a piece of wire just below red heat and press it into the plastic waste. If it penetrates the waste material, it is a thermoplastic; if it does not, it is a thermoset.
- The type of plastic can be identified by scratching it with a fingernail or from the flexibility of the material. However, these tests are not always reliable.
- Flotation test. This test can be used to disentangle larger quantities of mixed plastics, as well as to separate them from nonplastics. The test is also useful for making the complicated distinction between PP and HDPE, and between HDPE and LDPE. When placed in a tube of water and alcohol in certain proportions (this can be tested using a "hydrometer," with a density range of 0.9–1.0 g/cm^3) the materials will separate according to their density; one material will sink and the other will float.

For example, in a water–alcohol mixture with an exact density of 0.925 g/cm^3, the PP will float and HDPE will sink; in one with a density of 0.93, LDPE will float and HDPE will sink.

However, note that the flotation test is not exact enough to distinguish PP and LDPE, since their densities can overlap. In this case, the fingernail test and the visual appearance of the waste material may be more conclusive indicators.

Another flotation test using pure water and salt can be used to distinguish PS and PVC, both of which sink in pure water. When a specific amount of salt is added to the water, the PS will float to the surface, while the PVC and dirt will remain on the bottom of the container. The amount of salt may be determined by experience.

- *Burning test.* This test is carried out as follows. Cut a 5-cm-long piece of the plastic material, 1 cm wide at one end, and tapering to a point at the other.

Hold the sample over a sink or stone, and light the tapered end. The color and smell of the flame can be used to identify the type of polymer. PVC can be confirmed by touching the sample with a red-hot copper wire and returning the wire to the flame; it should burn with a green flame. Burn of all residue before repeating the test with the same wire. To facilitate identification, in the United States, the Society of Plastics Industry (SPI) has developed a model coding system (using numbers combined with the abbreviations PE, PP, etc.), which is now also being introduced in Europe (see Fig. 8). This coding system is especially suitable for molded products where the coding can be engraved onto the molds. In this way, households will be able to identify and separate the various types of plastics before disposal.[107,108]

4.4.3.4. Size reduction

Size reduction techniques such as cutting, shredding, and agglomeration of the waste plastic articles serve to increase the density of the material. This "densification" helps to reduce transport costs, and the smaller pieces can be more easily fed into further reprocessing machines. There are varieties of the techniques and procedures that can be applied.[111]

4.4.3.4.1. *Cutting*

The first step in the process of material transformation involves cutting up the waste plastic materials into smaller pieces. This is needed for items such as jerry cans and buckets, which are too large to fit into the hopper of the shredder. These items can be cut first with a circular saw or with a band saw.

4.4.3.4.2. *Shredding*

The raw waste materials to be fed into a shredder, if necessary cut into pieces beforehand, should consist of clean objects and be selected according to product form, polymer type, and color. These materials are fed into the hopper on top of the shredder, as shown in Fig. 9. The rotating cutting blades then shred the waste materials. When the pieces are small enough, they fall through a grid into the tray placed to the right of the machine.

Figure 8. The American SPI Coding System. PETE: polyethylene terephthalate; HDPE: high-density polyethylene; V = PVC: polyvinyl chloride; LDPE: low-density polyethylene; PP: polypropylene; PS: polystyrene; Other: Other polymers or blends.

The final products of shredding process are irregularly shaped pieces of plastics that can then be sold to reprocessing industries and workshops. Depending on the quality and type of raw material, and the desired quality of the final product, different types of plastic waste may be mixed to a certain extent.[111]

4.4.3.4.3. *Agglomeration*

It is not advisable to feed soft plastic waste, such as bags and plastic sheets, directly into a shredder or extruder. Preferably, an agglomerator should be used to cut, preheat (or pre-plasticize), and dry these waste plastics. Agglomeration improves the quality of the final product. In addition, it will increase the density of the material, which results in a more continuous flow of material in the extruder and thus, an increase of efficiency.

The materials fed into the agglomerator should be clean. Figure 10 shows an agglomerator. In such a machine, mechanical energy produced by the rotation of the cutting blade at high speed is transformed into heat through friction. The bulk density of the raw materials in the agglomerator increases through shrinkage and

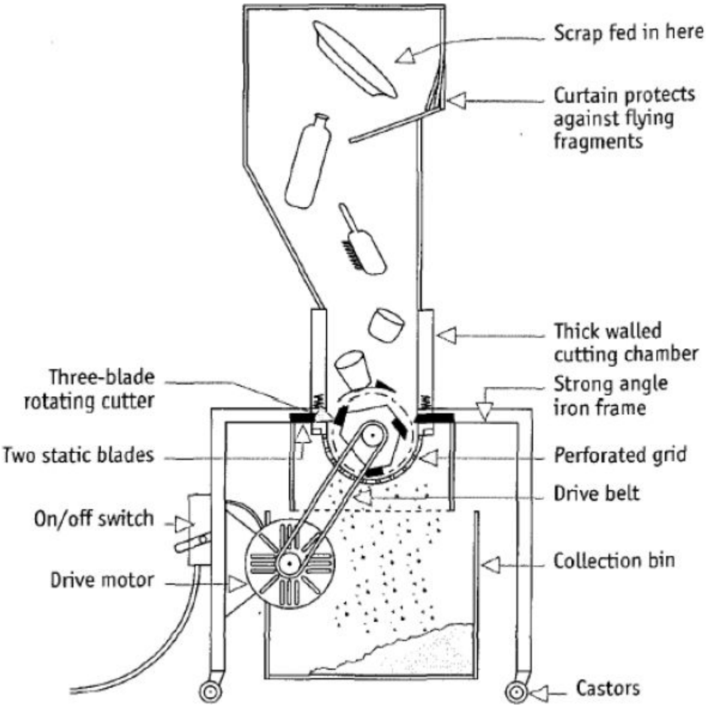


Figure 9. Shredding Machine.

partial plasticization. When the material is cooled rapidly at this stage, it solidifies as it is being cut, resulting in coarse, irregularly shaped grains, often called crumbs.

Additional heating is provided by a heating element fitted around the agglomerator or by the introduction of steam into the agglomerator. The rapid cooling can be achieved by adding a cup of water or by creating an outward current of air. Sometimes, the resulting crumbs are then sieved to remove any fine particles.

4.4.3.5. Further reprocessing techniques

Pelletizing and product manufacturing are the final steps in the plastics recycling process. These processes require that the waste plastics have first been sorted according to polymer type and that they have been cut into small and uniformly sized pieces. Shredded and agglomerated materials can be used directly for product manufacturing processes. In other words, they may not pelletized before reprocessing, although this is not usually done. Normally, the shredded and agglomerated waste plastics are firstly pelletized. By this way, the quality of the molded final products can be improved. The use of pellets also increases the efficiency of the product manufacturing process, due to the lower bulk density of shredded and agglomerated waste plastics compared to pellets.

The most common molding processes employed in reprocessing industry are extrusion, injection molding, and blow molding. Film blowing is used in the manufacture of plastic bags.[112]

Figure 10. Agglomerator.

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Chapter 7

REUSE OF FOUNDRY SAND AS CONSTRUCTION MATERIAL

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Abstract

The ultimate solution to the high volume of excess foundry sands generated in metal casting facilities is to beneficially reuse them. A variety of possible reuse programs suit the utilization of foundry sands. These programs mainly relate to civil and agricultural applications, e.g., highway embankment, concrete, backfills, topsoil, and growing amendments. Foundry sand exhibits qualified physical, mechanical, or chemical properties for these end-use products and is acknowledged as a marketable resource. Most often, foundry sand is characterized to match a target material, which has been adopted as a standardized component in an end-use product, e.g. fine aggregate in concrete. If the match gives favorable outcomes, the foundary sand is introduced fully or partially replacing the target material to form a technically competitive end-use product, e.g. foundry sand used in concrete. In addition to its engineering qualifications, foundry sand's environmental safety is also ensured to comply with regulations. The chemical characterization of foundry sand needs to be accomplished to demonstrate that it will not pose a thread to environments and human health during reuse programs. This chapter is organized to present the system (foundry) sand operation, foundry sand generation and management, chemical characterization of foundry sand, reuse applications of foundry sand.

Keywords: Foundry sand, reuse, construction materials, chemical characterization, censored data, engineering behavior, agricultural applications, disposals, metals, organics, toxicity, embankment, concrete, flowable fill.

1. Introduction

1.1. *Background*

One of the most pressing problems facing metal casting industry today is the disposal of foundry sand, a major solid byproduct generated in the casting processes of a foundry establishment. The US annual generation of foundry sand is reported to range from 6 to 10 million tons in 1999.[1] Disposal of foundry sand incurs a large amount of cost for foundry industry, whether disposal occurs in company-owned facilities or in municipal or privately owned landfills. Given the national average tipping fee of foundry sand to landfills at 15–75 USD/t,[2] there is a potential average annual savings of around 180 million USD on disposing expense with a 50% reutilization of foundry sand. Therefore, diversion of foundry sand into the streams of beneficial reuse will have a substantial positive impact on foundries' bottom lines.

The reuse of foundry sand offers substantial environmental and economic benefits. The cost and environmental impact of exploring natural resources, e.g., fine aggregate and soil, are mitigated by the substitutes of foundry sand for these materials. The cost of disposing foundry products to landfill sites will be saved. Both natural mineral resources and the urban environment are protected. The reuse program will result in substantial energy and cost savings for both foundries and user industries, as well as provide an environmental benefit at the local and national levels.

The critical qualifications of foundry sand for reuse alternatives include its environmental safe attribute and its technical suitability for reuse applications. The foundry sand is often arbitrarily regarded as a "dirty" waste by the public. It was presumed that the reuse of foundry sand would possibly pose threats to the environment and human bodies. Therefore, a sound understanding of chemical characterization of foundry sand helps eliminate the concerns and ensure environmental compliance. The physical and mechanical properties of foundry sand are necessary in matching foundry sand to appropriate reuse programs. The engineering behaviors of foundry sand based products are critical in validating the reuse of foundry sand and helping standardize products' technical specifications.

1.2. *Objectives*

This section focuses on the state-of-the-art foundry sand reuse programs, and is organized into four parts: (i) the introduction to operations of the foundry industry,

(ii) the chemical characterization of foundry sand, (iii) the reuse of foundry sand to various applications, and (iv) case studies. The first part is dedicated to the foundry operation introduction, including the general workflow in a foundry workshop, the casting processes, the types of binder and bonding processes, and the waste stream generations. The second part summarizes the chemical characterization related to foundry sand, including the current regulations and analytical methods, the metal and organic contaminants at total and leached levels, the variation of contaminants with leaching protocols, field/laboratory correlations, and contaminant extractability. The third part collects a number of accomplished reuse programs and categorized them in terms of end-use applications or products. Each reuse program contains the description of the principle or specification, the mechanism of foundry sand suitability for the reuse program, and the selected research efforts involving foundry sand reuse. The last part presents three case studies of reusing foundry sands in civil applications.

This chapter intends to be a referable contribution about the reuse of foundry sand. It aims at verifying that foundry sand is an environmental safe, economic competitive, and technically advantageous reusable resource. Its broad dissemination may prompt the acceptance of reusing foundry sand by more professionals, decision makers, and legal entities. It will also encourage foundry sand reuse in wider disciplines.

2. Foundry Industry Operations

2.1. *Introduction*

The scope of the metal casting industry encompasses a major segment of the US economy. Its sales reached 18.3 billion USD in 2001 (Fig. 1). Its shipments reached 12.2 million tons in 2001 (Fig. 2). It employs directly and indirectly 225,000 people in approximately 2,950 foundries. Metal casting is largely a small business industry. About 80% of US metal casters employ fewer than 100 people. About 14% employ 100–250 people and 6% employ more than 250. The industry's products, castings, enter into every field in which metals serve man, including automotive and light truck (35%), pipe and fittings (15%), construction, mining, and oilfield machinery (6%), internal combustion engines (5%), railroad (5%), valves (5%), farm equipment (3%), municipal castings (3%), and other (23%).

A metal-casting foundry is a commercial establishment for producing metal castings. A metal casting is a shape obtained by pouring liquid metal into a mold or cavity and allowing it to freeze and thus to take the form of the mold (Fig. 3). In terms of the manner of molding and core making process, metal-casting processes generally include sand-casting process, permanent-mold process, die-casting process, centrifugal process, shell-mold process, investment process, lost foam

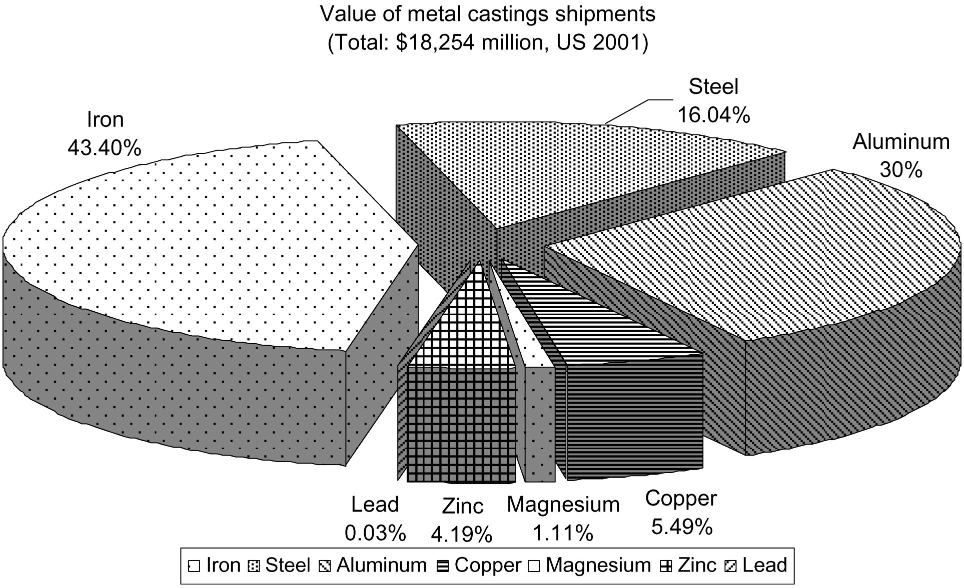

Figure 1. Value of Metal Casting Shipments.[3,4]

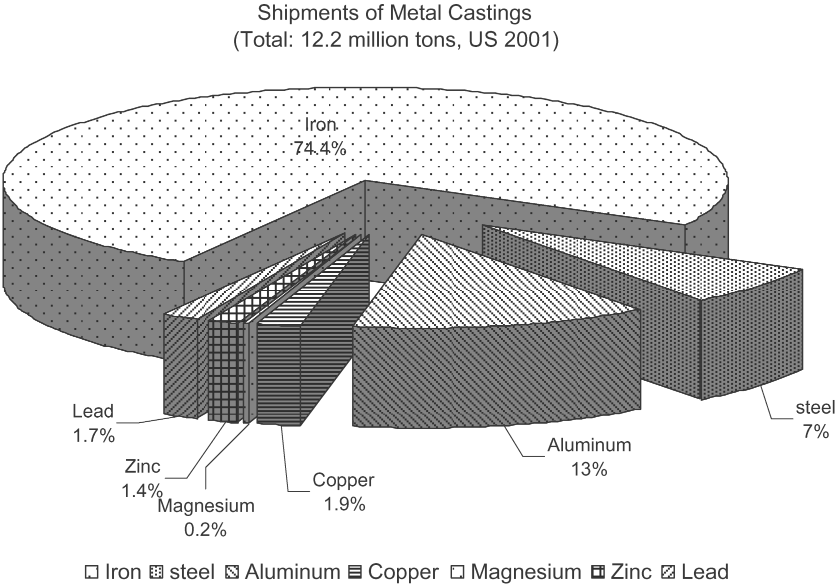

Figure 2. Producers' Shipments of Metal Castings.[3,4]

Figure 3. The Casting Process in a Foundry Facility.

process, plaster process, and graphite process. Sand casting process is the most widely-adopted casting process of low cost, high efficiency, and involving sand reclaimable cycles. It employs sand aggregate to make molds and cores.

In a typical foundry facility system, five fundamental steps are followed to fulfill the casting: patternmaking, core making, molding, melting and pouring, cleaning and reclaiming. The sequence of elements in such a system is shown in Fig. 4.

The metal casting industry is the "original recycler." Rather than use new or "virgin" materials as melt stock, nearly all foundries reuse scrap metals as their primary melting material. Annually, foundries consume 15–20 million tons of recycled scrap metal, giving new life to products that would otherwise go to landfill sites. The properties of casting metals can be improved with respect to fluidity, melting point, strength, and hardness by the addition of alloying elements.[6] The most common alloys used in metal casting are shown in Table 1.

The charge materials for typical iron casting consist of pig iron, iron and steel scrap, foundry returns, lime, and iron ore. Typical melting stock for steel casting includes pig iron, steel, cast iron, and alloying additions. The alloying elements commonly employed for the purpose of securing steel hardenability include manganese, chromium, molybdenum, nickel, silicon, copper, and vanadium, totally up to 87% by mass. The common metal charge of aluminum-based facility consists of clean foundry scrap (remelt), prealloyed aluminum pig and scrap of heterogeneous origin. Alloying elements, such as copper, silicon, zinc, nickel, chromium, titanium

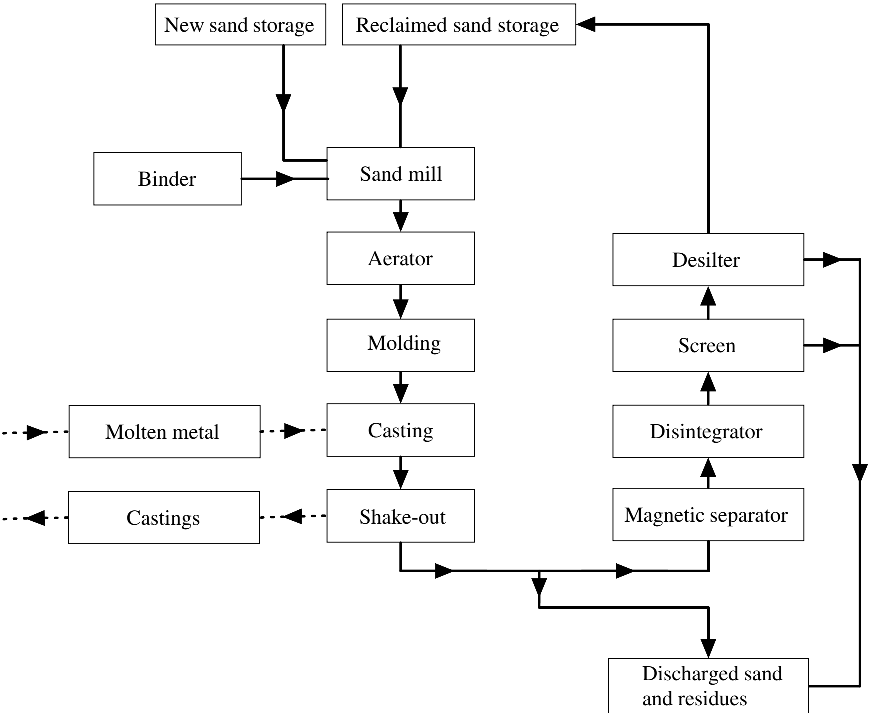

Figure 4. Flow Diagram of a Typical Molding Sand System.[5]

Table 1. Classification of Foundry Alloys.

Ferrous Castings		
Steel Castings	Iron Castings	Nonferrous Castings
• Plain carbon steel • Low-alloy steel • High-alloy steel	• Gray cast iron (high-carbon) • Ductile or nodular cast iron (spheroidal-graphite) • White cast iron (medium-carbon) • Malleable iron (annealed white iron with graphite)	• Aluminum-base • Copper-base • Lead-base • Magnesium-base • Nickel-base • Tin-base • Zinc-base • Miscellaneous

and manganese, are added as rich alloys or hardeners. The general metal charge for copper-based facility consists of clean foundry scrap (remelt) and pig or ingot of the desired analysis. Additions, such as zinc, tin, and lead, are made to achieve the desired analysis. Many copper-alloy castings require excellent surface finish as cast. Fines aggregate for moldings has been found to give a more desirable finish.

Facing materials, such as cereal, flour, plumbago, and sea coal, are dusted on the mold-cavity face to improve surface finish.

2.2. *Sand*

Sand is the most widely-used refractory aggregate for molds and cores making because it has the ability to absorb and transmit heat as it allows gases evolved during binder breakdown to pass between the grains, the ability of grains holding together and giving strength, and the ability to withstand high heat with moderate breaking down or fusing. As shown in Table 2, there are many types of sand utilized by the foundry industry. However, because of its wide availability and relatively low cost, silica sand is the one that makes most of the molding. Quartz and other silica minerals are the main components of silica sand, which is essentially repetitive interconnection of SiO_2. Silica sand is found in nature on the bottoms and banks of rivers, lakes, on seashores, in dry river deposits, and in substrata layers of the earth. Many of these deposits are mined and used without alteration. Sometimes, to obtain a mechanical strength, manufactured sand is used in the casting systems.

Sand employed in a mechanical sand preparation and handling system, i.e., sand mill in Fig. 4, is called system sand. In a sand casting process, system sand is used to prepare molds and cores prior to pouring melting metals. The mold is made by packing system sand around a pattern (Fig. 5). When the pattern is withdrawn, its imprint provides the mold cavity. Cores are forms, usually made of system sand, which are placed into a mold cavity to form the interior surface of a casting. The void space between the core and mold-cavity surface is what eventually becomes the casting. Cores are made by core-boxes, which are filled with core sand, rammed, and struck off. For examples, a core may be a simple round cylinder form needed to core a hole through a hub of a wheel or bushing. The external shape of a core becomes the internal shape of the casting. Molds are made by various bonding systems, including green sand, organic bonding systems, and inorganic bonding systems (see Sec. 2.3). Cores are usually made of organic and inorganic bonding systems. In general, sand molds are shaken off and reclaimed to new molds or cores.

2.3. *Sand Binders and Bonding Systems*

A binder is a material added to the sand or provided by nature with the sand that imparts cohesiveness. Its typical mass percentage varies from 0.5% to 15% based on the base system sand, binder, and bonding system. Binders are categorized into three groups; they are clay binders, organic binders, and inorganic binders. The last two groups are frequently called chemical binders, which encompass a wide range

Table 2. Comparison of Foundry Base Sands.[7]

	Silica	Olivine	Chromite	Zircon	Zircon/ Aluminum Silicate	Staurolite	Carbon Sand
Composition	SiO_4^{4-} tetrahedron connections, >95% SiO_2	SiO_4^{4-} tetrahedron by Mg^{2+} and Fe^{2+}, SiO_2 (42%), MgO (50%), Fe_2O_3 (7%)	SiO_2 (2%), MgO (10%), Fe_2O_3 (25%), Cr_2O_3 (50%), Al_2O_3 (15%)	SiO_2 (32%), ZrO_2 (65%)	—	—	Petroleum, C
Color	White-light, brown	Greenish gray	Black	White-brown	—	Dark brown	—
Hardness	6.0–7.0	6.5–7.0	5.5–7.0	7.0–7.5	6.5–7.0	6.5–7.0	—
Dry bulk density, lb/yd^3 (kg/m^3)	2294–2696 (1362–1600)	2696–3370 (1600–2000)	4179–4455 (2480–2644)	4320–4996 (2564–2965)	4186–4536 (2484–2692)	3862–3943 (2292–2340)	1729 (1026)
Specific gravity	2.2–2.6	3.2–3.6	4.3–4.5	4.4–4.7	3.2–4.0	3.1–3.8	—
Grain shape	Angular/rounded	Angular	Angular	Rounded/angular	Rounded	Rounded	Rounded
Thermal expan., 1,600° (cm/cm)	0.018	0.0083	0.005	0.003	0.005	0.007	—
Appar. heat transfer	Average	Low	Very high	High	High	High	
Fusion point, °F (°C)	2,600–3,200 (1427–1760)	2,800–3,200 (1538–1760)	3,200–3,600 (1760–1982)	3,700–4,000 (2038–2204)	3,300–3,600 (1816–1982)	2,500–2,800 (1371–1538)	>4,532 (>2500)
High temp. reaction	Acid	Basic	Basic	Acid	Slightly acid	Slightly acid	—
Wettability with molten metal	Easily	Not generally	Resistant	Resistant	Resistant	Resistant	—
Chemical reaction	Acid-neutral	Basic	Basic-neutral	Acid-neutral	Neutral	Neutral	—
AFS GFN	25–180	40–160	50–90	95–160	Approx. 80	Approx. 70	—
Cost: 1 cheapest; 4 most expensive	1	2	3	4	—	—	3
Density: 1 lightest; 4 heaviest	1	2	3	4	—	—	1

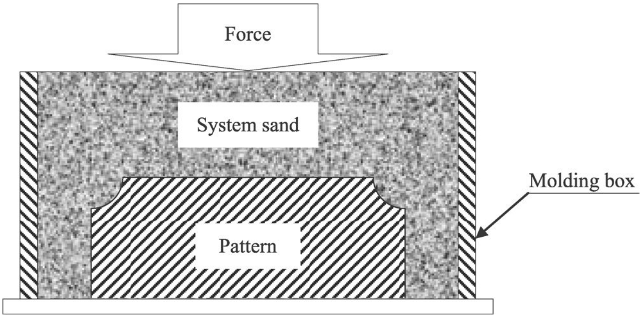

Figure 5. Molding in a Green Sand System.

of chemicals, e.g., urethane, phenol, furfural alcohol, and sodium silicate. Clay is uniquely used for green sand system. Organic and inorganic binders are used for chemical bonding system. The bonding strength of a binder is activated by additives (e.g. water) or external forces (e.g. heat and gas) to bond the system sand together into a mold or core in a bonding system. According to the bonding activation mechanism, the bonding system can be classified into a variety of types, e.g., cold-box, warm-box, hot-box, and no-bake. Combinations of binders and bonding systems are presented in Table 3.

In addition to binders, one or multiple additives are added to the system sands to improve some special bonding or casting features, or catalyze the binders. Examples include sea coal, wood flour, and silica flour.

2.3.1. *Clay and Green Sand*

The mold-making process based on clay is called green sand, or the green sand system. In the bonding process of green sand, clay is added as a binder in the amount of 4–10% by weight of base sand, and is used to make small, medium and large molds. The production of the green sand system is relatively inexpensive and provides favorably bonding strength of the molds. The major subdivisions of the clay family are montmorillonite (mostly bentonite), kaolinite, and illite. Clay is chemically a hydrous alumina silicate, which is able to provide cohesion and plasticity in the green state and also high strength when dried. Most of the cases, two types of bentonite, sodium bentonite (western-type) and calcium bentonite (southern-type), are blended to produce strong molding sands with low moisture content.

Green sand is the most widely used bonding process worldwide because it is low in cost, high in performance, environmentally benign, and the materials are reusable. Green sand is composed of uniform high-quality refractory sand (85–95%), cohesive bentonite clay (4–10%) as the binder, a volatile carbonaceous sea-coal additive (2–10%) to improve casting surface finish, iron oxide (0.5–5%) for strength, and

Table 3. Conventional Binders and Bonding Systems.[1]

	Thermosetting Processes	Self-setting Processes	Gas-cured Processes
Organic Binders	A. Shell process: Novolac (shell) resin + Phenol formaldehyde B. Hot-box processes: Urea formaldehyde Phenol formaldehyde a. Novolac b. Resole Furan modified a. UF + FA b. PF + FA c. PF + UF C. Warm-box process: Furan a. Free formaldehyde b. UF or PF c. Sulfonic acids or copper salts D. Core oil process: Oils Water activated cereal	A. Acid no-bake processes: Furan no-bake a. H_3PO_4 b. TSA c. BSA Phenol formaldehyde a. TSA b. BSA c. Xelenesulfonic acid B. Ester cured processes: Phenolic resole a. Free phenol b. Free formaldehyde C. Urethane no-bake (amine cured) processes: Alkyd urethane a. Vegetable oil b. Polyisocyanate Phenolic urethane a. Pyridine derivative b. Polyphenyl PIC Polyol Urethane	A. Free radical curing: Vinyl urethane oligomer a. $N_2 + SO_2$ b. Epoxy/ hydroperoxyde B. Cold-box processes: Phenolic urethane Polymeric isocyanate a. TEA vapor + air b. DMEA vapor + air Furan + SO_2 a. Methyl alcohol b. MEK peroxide Acrylic/epoxy + SO_2 a. Hydroperoxide Phenolic resole + ester a. Glycol ethers b. Methylformate vapor
Inorganic Binders	A. Clay based processes: Bentonites Fire clays Kaolinite	A. Ester cured processes: Sodium silicate a. Glycerol diacetate b. EGDA c. glycerol triacetate Ethyl Silicate B. Cement bonding process: Hydraulic cements C. Oxide cured process: Phosphates a. Aluminum phosphate b. Magnesium oxide	A. CO_2 silicate process: Sodium silicate (SiO_2:Na_2O) + CO_2

Notes: FA = furfural alcohol; PIC = polyisocyanate; UF = urea formaldehyde; TSA = toluenesulfonic acid; PF = phenol formaldehyde; BSA = benzenesulfonic acid; TEA = triethylamine; MEK = methyl ethyl ketone; DMEA = dimethylethylamine; EGDA = ethylene glycol diacetate water (2–5%).[1] It is called “green” sand, not for its color, but for the contained moisture contents which distinguish it from dry sand.

The principal properties of green sand are governed by the clay–water–silica relationship. Because of the lack of electrical neutrality of clay and an unequal balance of charge at the fractured quartz surface, polarized water molecules are attracted either between the clay plates, or to quartz surface. As a result, a linkage, quartz–water–clay–water–quartz (or clay), is, therefore, established throughout the molding sand. Besides the electrostatic bonding, surface tension forces, and interparticle friction bond also build the clay bonding system.

2.3.2. *Chemical Binders and Chemical Bonding Processes*

Chemical binders (organic and inorganic) are usually used for small-medium size molds and cores, relatively stronger and good erosion resistance. Organic binders include two major groups: resin and oil-based binders. They are combustible, and destroyed by heat in the casting processes. The resin-based binders are made by mixing various proportions of phenolic, furfural alcohol, urethane, and formaldehydes, ranging from 1.5% to 8% by weight of base sand. They are liquids or gums, natural or synthetic, while the individual molecules have the capacity to polymerize or fuse together to form very long chains. This polymerizing reaction can be triggered by certain chemical additives or heat. The resulted chain hardens to form a cohesive bond in the system sand matrix. The other organic binder is oil-based binder. Oil-based binders, in the amount of 0.5–3%, can be either natural oils (e.g., linseed, perilla, tung, and dehydrated castor oils) or manufactured oils (e.g., unsaturated mineral oils, synthetic oil, and alkyd resin). Their hardening/bonding mechanism is similar to resin, i.e., polymerization with or without heat.

Inorganic binders include sodium silicate and Portland cement, which are incombustible. Compared with organic binders, they are environmental safe, low cost, low sand sensitivity, and low gas evolution. Sodium silicate, generally called water glass, refers to a three-ingredient system, consisting of silica, sodium oxide, and water. Sodium silicate is unique in that it can undergo four very distinct chemical reactions, including hydration/dehydration, gelation, precipitation, and surface charge modification, to bond other materials together. When Portland cement is mixed with water, its chemical compound constituents (tricalcium silicate, dicalcium silicate, tricalcium aluminate, etc.) undergo a series of chemical reactions that cause it to harden (or set) and bond other materials together. Both inorganic binders develop great hardness and strength by the hydraulic setting actions of sodium silicate or Portland cement. However, inorganic binders present relatively slow cure characteristics, and once used, it is poor to breakdown or reused.

Chemical binders need to be exposed to appropriate chemical bonding processes to activate their cohesion features. The chemical bonding processes are normally classified into three groups, defined according to the approaches to hardening, hot curing, cold setting, and gas or vapor hardening (Table 3). After bonding activations, the system sand can hold a cold and hot tensile strength ranging from 50 to 150 psi (34 to 102 kPa). The most widely-used bonding processes include cold-box, no-bake, hot-box, and warm-box (Table 3).

Cold-box core and mold making describes any binder process that uses a gas or vaporized catalyst to cure resin-coated sand while the sand is in contact with a room-temperature pattern. No-bake core and mold making process is designed around a continuous mixer that blends the binder with the catalysts or co-reactants as it coats the sand and dispenses the coated sand into the tooling. In the hot-box core making process, after the sand has been coated, it is blown into a heated core-box pattern at temperature of 450–550°F (230–290°C). It starts to harden and form a shell-like skin. Warm-box core making uses the same sand processing system as hot-box. They differentiate in the temperature of heating pattern. The typical temperature for warm-box pattern ranges from 350–450°F (180–230°C). The most employed combinations of binder types and bonding process are chronologically listed in Table 4.

2.4. *Waste Streams Generation and Management*

When liquid metal is poured into a mold, Carey[9] described that the following events occur: (a) Free moisture is driven off the sand beginning at the mold metal interface and either condenses in the cooler parts of the mold or evolves as steam into the atmosphere; (b) Volatile matters are driven from organic materials and from the residual binders adsorbed to the surfaces of core sand that has entered into the system by the shake-out route; (c) As the sand heats up the clay bonding materials are deactivated and "killed." At temperature above 1,100°F (600°C), combined water is driven out of the clay bond. All clays lose their combined water in parts of the molds as they reach 1,470°F (800°C), and become ineffective; (d) The sand heats up as the casting solidifies and expands; (e) When the sand is heated up to temperature over 1500°F (820°C), the possibility of chemical reaction between metal and sand, fusion of the sand, increases.

After pouring, casting, and shake-out operations, the bulk system sands are sent to reclamation units to recover them for another cycle and save virgin sand use. The principal methods of sand reclamation include dry scrubbing (pneumatic, mechanical, or shot blast) for clay, organic and inorganic binder system sand, calcination (thermal) for organic binder system sand, and wet washing for clay and silicate binder system sand. A fraction of excess foundry sand is generated as a waste stream in this step.

Table 4. Chemical Bonding Series.[8]

Commercial Intro.	Systems	Major Resin, Catalyst, and their Mass Percentage Based on Base Sand
1950	Core oil	Oil (up to 20 ingredients), cereal, water, less release agent, 2–3%
1950	Shell	Phenolic novolak flake resin, hexamethylenetetramine
1952	Silicate/CO_2	Sodium silicate, water, CO_2, 3–5%
1953	Airset oils	—
1958	Phenolic acid catalyzed no-bake	—
1958	Furfural alcohol acid catalyzed no-bake	—
1960	Furan no-bake	Furfural alcohol (from corn husks, rice hulls), 0.9–2.0%
1962	Phenolic/furfural alcohol hot-box	Phenol, furfural alcohol, urea, formaldehyde, ammonium chloride, ammonium nitrate, 1.5–2.0%
1965	Oil urethane no-bake	Oil resin, polymeric isocyanate, amine, and metallic compounds, 1–2%
1967	Phenolic CO_2 cold-box	—
1968	Phenolic/urethane/amine cold-box	Phenolic resin, polymeric isocyanate, amine, 0.8–1.5%
1968	Silicate ester catalyzed no-bake	Sodium silicate, ester catalyst, 2–4%
1970	Phenolic urethane no-bake	Phenolic polyol, isocyanate, amine, 0.8–1.75%
1977	Furfural alcohol SO_2	Phenolic modified furan, silane, organic peroxide reacted with SO_2
1978	Polyol urethane no-bake	Polyol resin, isocyanate, amine
1978	Furfural alcohol warm-box	20% less resin than that of hot-box, copper salts of toluene sulfonic acid, copper chloride
1980	CO_2 cured alkaline phenolic cold-box	Alkaline Phenolic resin, around 2.0–3.5%
1982	Free radical curing SO_2	Acrylic resin, SO_2, 0.5–2%
1983	Epoxy SO_2	Epoxy, SO_2, 0.5–2%
1984	Phenolic ester no-bake	Phenolic resole resin, 1.0–1.5%, ester (organic, alcohol)
1985	Phenolic ester cold-box	Water soluble, ester (methyl formate)
1992	Alumina phosphate	—

After a number of molding-casting-reclaiming-molding cycles, the system sand degrades physically and chemically. Reclamation tends to smooth and round the surface of the grains. Breakdown of quartz grains and accumulation of fusible fines reduce both permeability and refractoriness. The shake-out may lead to a mixture of mold and core sand with different gradation that adversely affects molding quality and impedes an efficient reclamation. Inorganic residuals in the pores of grains may not be fully removed, which results in molding incompatibility and casting defects. Even if the coating is almost completely removed during every reclamation cycle, the chemistry of the reclaimed sand, vary somewhat with each cycle.[10] The surface chemistry of reclaimed sand is drastically altered by the resin, catalyst, and additives. For green sand, the clay at or near the casting surface becomes heated and destructed to “dead.” After this has occurred, milling and additions of water will not bring back the bond. Too much “dead” clay results in brittle sand, poor strip, and bad lifts. Hence, impacts to system sands, in the repetitive mold/core making, casting, and reclamation processes, eventually impair sand qualifications as a refractory casting material.

Degraded system sand is no longer structurally suitable for molding and core making purposes and needs to be removed from the system to maintain the quality of casting. The removed system sand or foundry sand becomes one of foundry byproducts. An overwhelming amount of foundry sand, up to 10 million tons, is discarded from facilities annually, which occupies over 80% foundry byproducts by weight. Besides the system sand, a typical foundry facility can generate many other byproducts, e.g., copula slag, scrubber sludge, baghouse dust, shotblast fine, buffling waste, etc. These byproducts are classified into four waste streams depending upon their collection manners: sand, dust, slag, and sludge. Dust is mainly the fine particles collected by various collectors installed in a foundry facility. The formation of slag used in metal melting and refining processes can be summarized as protection of the melt from contaminations from the furnace atmosphere and combustion products, insulation of the melt to maintain the heat in it, and an acceptor of unwanted materials as refining media to the melt. Sludge is mainly the wet scrubbing collection. Table 5 presents the descriptions for waste streams when they are generated and managed. Foundry sand presents safer environmental characteristics than other waste streams, which put it a competitive advantage over slag and dust for its possible reuse alternatives.

Table 5. Descriptions of Foundry Waste Streams.

Waste Streams	Descriptions
Sand	Sand, green sand, mold, shell core, oil sand, furfural alcohol, recyclable sand, isocure, tank, tail, pulley, sand dropout
Dust	Dust, shot blast, shake-out (sand), muller, grinding, wheelabrators, collector, cleanroom, blast, fume, pretreater & desulfirazation, cutoff, fines, finish, pagborn, carter day collector, large wet collector, sand reclaimer, tailing, baghouse
Slag	Slag, furnace refractory, iron, ductile, melt, copula, popcorn, ladle relining, crucibles/brick
Sludge	Sludge, copula, furnace, pugmill, cake

3. Chemical Characterization of Foundry Sand

3.1. *Introduction*

The understanding of chemical characterization of foundry sand is a critical issue that needs to be addressed prior to its acceptance in beneficial reuse programs. Foundry sand may contain contaminants that possibly pose threats to the environments and the human health. Foundry sand's chemical characterization assists engineers in making a decision between beneficial reuse, reclamation or waste disposal. Chemical characterization spans a range of variables, including contaminant types, their quantification and distributions, and the environmental compliance assessment. The threshold values of contaminants are unique quantitative criteria against which the environmental concern of foundry sand is assessed. These threshold values are regulated in environmental regulations at federal, state, and local levels.

3.2. *Federal Act and Regulations*

Resource Conservation and Recovery Act (RCRA) was enacted in 1976 to address the issue of "protect human health and the environment from the improper handling of solid waste and to encourage the conservation of natural resources." US Environmental Protection Agency (US EPA) was authorized to administer the RCRA program, including managements of both hazardous waste and solid wastes. Subtitle C of RCRA addresses "cradle-to-grave" requirements for hazardous waste, namely its generation, transportation, treatment, storage, and disposal. Subtitle D of RCRA sets forth less restrictive framework for the management of nonhazardous solid waste. Foundry sand is still under RCRA's rules as a solid waste when it is removed from a foundry facility.

The Code of Federal Regulations (CFR) is a codification of the rules published in Federal Register by executive departments and agencies of the federal government. The CFR is divided into 50 titles which represent broad areas subject to federal regulation. Environmental protection regulations are contained mainly in Title 40. The CFR Title 40 contains the rules that US EPA uses to implement the requirements of the RCRA. The CFR parts closely relating to the characterization of foundry sand include Part 261 "Identification and Listing of Hazardous Waste." Part 141 "National Primary Drinking Water Regulations," Part 143 "National Secondary Drinking Water Regulations," and Part 264 "Standards for Owners and Operators of Hazardous Waste Treatment, Storage, and Disposal Facilities."

Part 261 identifies those solid wastes that are subject to regulation as hazardous wastes. This part specifies the maximum concentration of contaminants for the toxicity characteristic with regard to a hazardous waste. Toxic threshold values in Part 261 are generally used for assessing whether a solid waste is a hazardous material or not by implementing a US EPA toxicity leaching protocol. Any violation of a contaminant leads to a judgment of hazardous attribute of a solid waste.

Parts 141 and 143 establish safe drinking water regulations and limits pursuant to the Public Health Service Act, mainly for the protection of human health. Part 264 contains the maximum concentration of constituents that the uppermost aquifer underlying the waste management area meets for groundwater protection purpose. These water-related criteria apply to the total analysis of a waste sample, and may be referred to in some special cases, e.g., most stringent situations and drinking water receptors near to disposal sites. Threshold values of inorganic and organic chemicals specified in these CFR parts are summarized in Tables 6–8.

3.3. *Chemical Analytical Techniques*

The ability to accurately detect and quantify an element or compound present in a sample, even at extremely low concentration, is critically important in terms of a

Table 6. Threshold Values of Inorganic Chemicals for Safe Drinking Water and NonHazardous Material[11–13].

	Nonhazardous Material	Drinking Water	Groundwater
Chemicals (mg/L)	By TCLP	By Total Analysis	
Aluminum	—	0.05–0.2	—
Antimony	—	0.006	—
Arsenic	5.0	0.01	0.05
Barium	100.0	2.0	1.0
Beryllium	—	0.004	—
Cadmium	1.0	0.005	0.01
Chromium	5.0	0.1	0.05
Copper	—	1.0	—
Iron	—	0.3	—
Manganese	—	0.05	—
Mercury	0.2	0.002	0.002
Lead	5.0	0.015	0.05
Selenium	1.0	0.05	0.01
Silver	5.0	0.1	0.05
Thallium	—	0.002	—
Zinc	—	5.0	—
Reactive cyanide	250 (Total)	—	—
Reactive sulfide	500 (Total)	—	—
Ignitability	140 °F (60 °C)	—	—
pH for corrosivity	2–12.5 (Total)	—	—
Free cyanide	—	0.2	—
Fluoride	—	4.0	—
Nitrate	—	10.0	—
Nitrite	—	1.0	—

Table 7. Threshold Values of Organic Chemicals for Safe Drinking Water and NonHazardous Material.[11–13]

	Nonhazardous material	Drinking water	Groundwater
Chemicals (ug/L)	By TCLP	By Total Analysis	
Alachlor	—	2	—
Atrazine	—	3	—
Benzene	500	5	—
Benzo(a)pyrene	—	0.2	—
Carbofuran	—	40	—
Carbon tetrachloride	500	5	—
Chlordane	30	2	—
Chlorobenzene	100,000	100	—
Chloroform	6,000	—	—
o-Cresol	200,000	—	—
m-Cresol	200,000	—	—
p-Cresol	200,000	—	—
Cresol	200,000	—	—
2,4-D acid	10,000	70	100
Dalapon	—	200	—
1,2-Dibromo-3-chloropropane (DBCP)	—	0.2	—
o-Dichlorobenzene	—	600	—
p-Dichlorobenzene	7500	75	—
1,2-Dichloroethane	500	5	—
1,1-Dichloroethylene	700	7	—
cis-1, 2-Dichloroethylene	—	70	—
trans-1,2-Dichloroethylene	—	100	—
Dichloromethane	—	5	—
1,2-Dichloropropane	—	5	—
2,4-Dinitrotoluene	130		—
Di(2-ethylhexyl)adipate	—	400	—
Di(2-ethylhexyl)phthalate	—	6	—
Dinoseb	—	7	—
Dioxin (2,3,7,8-TCDD)	—	0.00003	—
Diquat	—	20	—

waste stream assessment for potential contamination. For instance, when monitoring disposed foundry sand, the toxicity violation of a single constituent in its leachate is often taken as evidence that the foundry sand has a direct impact on the environmental quality. The assessment as out-of-compliance could be extremely costly to the facility, and may prevent the beneficial reuse of generated byproducts.

3.3.1. *Total Analyses and Leaching Analyses*

Chemical analytical techniques are used to prepare, identify, and quantify inorganic and organic chemicals contained in foundry sands. The analyses are mainly classified

Table 8. Threshold Values of Organic Chemicals for Safe Drinking Water and NonHazardous Material (Cont.)[11–13]

	Nonhazardous material	Drinking water	Groundwater
Chemicals (ug/L)	By TCLP	By Total Analysis	
Endothall	—	100	—
Endrin	20	2	0.2
Ethylbenzene	—	700	—
Ethylene dibromide	—	0.05	—
Glyphosate	—	700	—
Heptachlor	8	0.4	—
Heptachlor epoxide	—	0.2	—
Hexachlorobenzene	130	1	—
Hexachlorobutadiene	500	—	—
Hexachloroethane	3,000	—	—
Hexachloropentadiene	—	50	—
Lindane	400	0.2	4
Methoxychlor	10,000	40	100
Methyl ethyl ketone	200,000	—	—
Nitrobenzene	2,000	—	—
Oxamyl (vydate)	—	200	—
Polychlorinated biphenyls (PCBs)	—	0.5	—
Pentrachlorophenol	100,000	1	—
Picloram	—	500	—
Pyridine	5,000	—	—
Simazine	—	4	—
Styrene	—	100	—
Tetrachloroethylene	700	5	—
Toluene	—	1,000	—
Toxaphene	500	3	5
2,4,5-TP (Silvex)	1,000	50	10
1,2,4-Trichlorobenzene	—	70	—
1,1,1-Trichloroethane	—	200	—
1,1,2-Trichloroethane	—	5	—
Trichloroethylene	500	5	—
2, 4, 5-Trichlorophenol	400,000	—	—
2, 4, 6-Trichlorophenol	2,000	—	—
2, 4, 5-TP (Silvex)	1,000	—	—
Vinyl chloride	200	2	—
Xylenes, total	—	10,000	—

into two types: total chemical analysis and leaching chemical analysis. Total chemical analysis determines the chemical mass percentage in dry or as received solid waste samples without any extraction. Leaching chemical analysis evaluates the chemical concentration level in the leachate extracted from dry or as received solid waste samples by using a leaching protocol. Lechate characterization is generally employed to determine whether the disposed solid wastes are hazardous or not. In the hazardous assessment of foundry sand in the United States, the most used protocols include toxicity characteristics leaching procedure (TCLP), synthetic precipitation leaching procedure (SPLP), and shake extraction of solid waste with water (ASTM D3987).

After the preparation of samples, e.g. to digest as-received or leachate samples into an acid solution, chemical procedures are used to measure an element or a compound in the samples. For foundry sand chemical characterization, the most used procedure manuals include "Test Methods for Evaluating Solid Waste, Physical/Chemical Methods,"[14] and "Methods for the Determination of Metals in Environmental Samples."[15] Out of the manuals, inorganic chemicals are commonly analyzed by using analytical chemistry techniques: atomic absorption spectrophotometry (AAS) and inductively coupled plasma-mass spectrometry (ICP-MS). The organic chemicals are usually measured by following methods: gas chromatography (GC), gas chromatography-mass spectrometry (GC-MS), and high-performance liquid chromatography (HPLC).

3.3.2. *Toxicity Characteristic Leaching Procedure (TCLP)*

TCLP is a leaching protocol released by US EPA as Method 1311 in source.[14] This leaching protocol is meant to model the leaching behavior of a material disposed in an actively decomposing municipal solid waste landfill in which carboxylic acids are formed from microbial processes, which is a particular worst-case scenario under RCRA specification of mismanagement scenario of wastes. If disposal conditions are different from the municipal landfill conditions, it may be better to involve alternative protocols to predict and provide better numerical estimates of the leaching of a waste. In the TCLP, solid samples are extracted with an acetate buffer solution. The extraction fluid employed is a function of the alkalinity of the solid phase of the waste. A liquid-to-solid ratio of 20:1 by weight is used for an extraction period of 18 ± 2 h. After extraction the solids are filtered through a 0.6–0.8-μm filter from the liquid extract, analyses are conducted on the leachate to determine the elements concentrations.

3.3.3. *Synthetic Precipitation Leaching Procedure (SPLP)*

SPLP is a leaching protocol released by US EPA as Method 1312 in source.[14] This leaching protocol is developed to simulate leaching through an industrial waste monofill under acid rain condition. The procedure is similar to TCLP; however, the amount of acidity used in the test is significantly less. Furthermore, an aqueous solution of nitric/sulfuric acid mixture is used in the SPLP as an extraction fluid, unlike a more aggressive buffered acetic acid in the TCLP. This procedure simulates more closely the field conditions for beneficial use in the general construction industry, where extractions by rainfalls and groundwater occur.

3.3.4. *Shake Extraction of Solid Waste with Water (ASTM D3987)*

ASTM D3987 is a neutral leaching protocol released by American Society of Testing Method (ASTM) as a testing standard.[16] This procedure is a useful indicator in considering the potential environmental impact of foundry sand waste and its beneficial use practices. The intent of this test method is that the water extraction simulates conditions where the solid waste is the dominant factor in determining the pH of the extract. This leaching protocol uses Type IV reagent water as the extraction solution, which causes this protocol less severe than above acidic medium leaching protocols (TCLP and SPLP).

Table 9 lists the major operation criteria of the three protocols that include extraction fluid types and pH values, liquid-to-solid ratios, extraction modes and periods, and analytes. The extraction fluid, specifically its pH value, differentiates these three protocols. TCLP is thought the most aggressive than the other two. The difference plays an important role in the extractability of chemicals. Therefore, selection of leaching protocol is decisive in the assessment of any solid waste. Each protocol is to model a particular extraction scenario. The mismatch between employed protocol and actual extractions leads to excessively conservative or unsafe assessment. The conservative conduction will take unnecessary cleanup expenses and lose the opportunity of a beneficial resource, whereas the unsafe conclusion may cause a threat to the human health and the environment.

Table 9. Operation Criteria of TCLP, SPLP, and ASTM D3987.

Operation Criteria	TCLP	SPLP	ASTM D3987
Extraction fluid	Acetic acid solution	Sulfuric/nitric acid solution	Reagent water
pH value of extraction fluid	Nonalkaline materials: 4.93±0.05, alkaline materials: 2.88±0.05	East of Mississippi River: 4.20, west of Mississippi River: 5.00	N/A
Extraction fluid and mode for volatile analytes	pH 4.93, zero-headspace extraction vessel	Deionized water, zero-headspace extraction vessel	N/A
Particle size reduction	<9.5 mm	<9.5 mm	Not required
Liquid-to-solid ratio	20:1 (by mass)	20:1 (by mass)	20:1 (volume:mass)
Extraction mode	Rotary agitation	Rotary agitation	Rotary agitation
Extraction period	18±2 h at 30 rpm	18±2 h at 30 rpm	18±0.25 h at 30 rpm
Leachate-solid separation	Glass fiber filters of pore size 0.6–0.8 μm	Glass fiber filters of pore size 0.6–0.8 μm	Glass fiber filters of pore size 0.45 μm
Collection time	One	One	One
Operation temperature	Room temperature	Room temperature	Room temperature
Analytes	Inorganic and organic analytes	Inorganic and organic analytes	Inorganic analytes

3.4. *Correlations between Total and Leaching Analyses*

A dilution at liquid-to-solid mass ratio 20:1 is included in TCLP, SPLP, and ASTM D3987 (Table 9). For instance, a 100-g sample of foundry sand needs to be diluted to a total 2000-mL solution using a volume of extraction fluid. The liquid-to-solid mass ratio provides an opportunity to correlate total and leaching analyses results. If 100 mg/kg arsenic is detected in the foundry sand sample by running a total analysis and 100% arsenic is extracted in the leaching analysis, arsenic concentration in the leachate would be [100(mg/kg) × 100(g)]/2000(mL) = 5 mg/L, equally one-twentieth of the total result 100 mg/kg, in number. Chemical extraction ratio is generally less than 100%. That is, the leaching results will numerically (ignoring units) be less than one-twentieth of the total results, e.g. less than $(1/20) \times 100 = 5$. Hence, although environmental impact or toxicity of a solid waste such as foundry sand currently is assessed in terms of its leaching characterization, the total characterization of the waste may implicate its worst leaching characterization by taking the dilution ratio into account. Remember to correlate respective units, e.g., mg/kg vs. mg/L.

Foundry sand samples are often pre-exposed to total chemical analyses to decide whether or not the samples require a leaching program. Running a total chemical analysis is much cheaper than running a leaching analysis. Based on the total chemical results and using the dilution ratios, the worst leaching characterization of foundry sand samples can be estimated. There is a possibility that a leaching program is not mandatory and expenses are saved, when the worst leaching characterization still indicates environmental safe results.

Currently, toxicity threshold values are uniquely specified for TCLP results. Lack of threshold criteria for other leaching protocols apparently becomes a barrier to assess the toxic characterization of samples that are exposed to SPLP, ASTM D3987, or other protocols. However, through exploring why and how to determine the TCLP toxicity criteria, the barrier may be eliminated.

A groundwater transport model (Fig. 6) is generally used to set toxicity threshold values against which concentrations in the TCLP leachate are compared. Toxicity threshold values represent back-calculation results from health-acceptable chronic exposure levels in a receptor well, through the unsaturated and saturated zones, back to the sources (i.e., the bottom of the landfill), accounting for the dilution and attenuation that is predicted to occur along the transport distance between the landfill bottom and the receptor well. US EPA assumes that water wells were situated 500 ft (150 m) down gradient from the landfill on which US EPA based its determination of dilution and attenuation factor of 100 for all of the listed toxic constituents based on the availability of chronic toxicity reference levels.[18] One of the available references is the standard of safe national drinking water and groundwater shown in Tables 6–8.

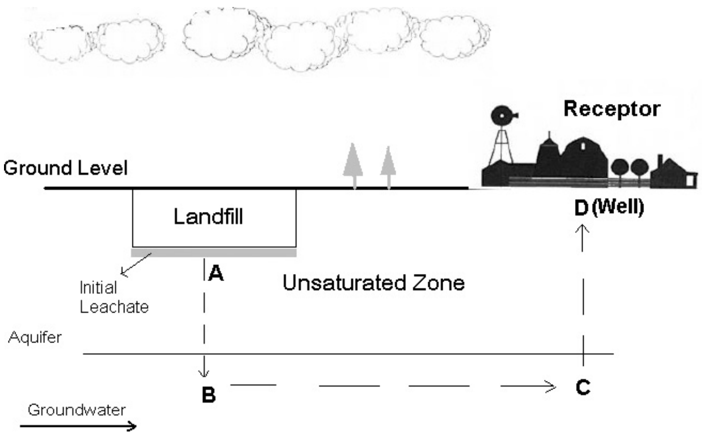

Figure 6. Schematic of Leachate Transport Model.[18]

For instance, the limit for mercury in drinking water and groundwater is 0.002 mg/L. Considering the dilution and attenuation occurred to mercury when it transports 500 ft (150 m) from the landfill leaching site to water wells or receptors, the maximum allowable concentration for mercury in the leachate ("A" in Fig. 6) is 0.002(mg/L) × 100 = 0.2 mg/L. This also partly explains why TCLP criteria are frequently 100 times of drinking and groundwater standards.

Based on the leachate transport (dilution and attenuation) model, the toxicity criteria are actually determined independent of leaching protocols. The threshold values are basically set with an aim to regulate the landfill bottom leachates ("A" in Fig. 6). In other words, any leachate needs to comply with the threshold values, regardless of how the leachate is extracted from the landfill. Leaching protocol type is not a factor of consideration when the toxicity criteria are concerned. TCLP is developed merely as an effort to simulate the leaching process occurred in the municipal landfills. The monofill landfill leachate, simulated by running SPLP, still needs to comply with criteria set at "A." Hence, TCLP toxicity criteria are actually applicable to the results of any other leaching protocol.

3.5. *Existence of Chemicals*

One of the important aspects related to the chemical characterization, particularly chemicals extractability, of foundry sand is the distribution or existing forms of chemicals in foundry sand. When chemicals are introduced into foundry sands, they may be distributed in one or more of the following forms: (a) present in the structure of foundry sand, clay, and organic binder; (b) specifically adsorbed to foundry sand, clay constituents, and organic binders; (c) dissolved in foundry sand solution (pore water); and (d) precipitated as pure or mixed solids. The first fraction generally exists if chemicals, specifically metals, occur in virgin sands or are fused to constituents under high-temperature and high-pressure conditions in casting processes. Electrostatic attractions by occupying exchange ions also lead to this form of existence. The last fraction is generally caused by the free existence of casting metals or discomposed binders. The mid-two fractions are modulated by the chemical concentrations. For simplicity, the chemicals forms in foundry sands can be grouped into four phases: (i) crystallized phase, representing the first form; (ii) adsorbed phase, representing the second form; (iii) aqueous phase, representing the third form; and (iv) solid phase, representing the forth form.

The crystallized phase represents chemicals embedded in minerals or organics by bond forces. The potential of extracting these chemicals out is relatively low. If this phase is the main existing form of chemicals, the volume of extracted chemicals will be low as well. In this situation, to change extraction fluid pH values may not make much difference, depending upon the chemical bond forces.

The adsorbed phase represents the accumulation of chemical ions at the interface between solid constituents and the aqueous phase. This phase is associated with the surfaces of solid constituents such as sand minerals, organic matter, and iron and manganese oxides. Specific surface magnitudes thus affect the amount of chemical accumulation. The main bonding forces include electrostatic attraction, van der Waals attraction, and cohesion of clay or organics.

The aqueous phase represents the chemicals existing in foundry sand solution, although least, as free (uncomplexed) ions (e.g. Cd^{2+}, Ni^{2+}). The potential for migration (or mobility) of this phase chemicals will be higher if higher amount of chemicals are present in the aqueous phase, even without the extraction of acid fluids. This is because chemicals in the aqueous phase are readily transported by physical and chemical processes such as advection, dispersion, and diffusion.

The solid phase represents simply chemicals precipitated in foundry sand matrix or foreign minerals/aggregates isolated from the three phases of foundry sand. These precipitates exist in three-dimensional solid phase and may consist of pure solids (e.g. Fe_2O_3, $CdCO_3$) or mixed solids. The solid phase may be present in foundry sand when chemical concentrations are significantly higher than their solubility limits or when pure chemical aggregates enter foundry sand matrixes. The chemicals that exist in solid phase are relatively immobile, but possibly highly extractable under acid extraction conditions and result in high concentrations in leachates.

The existence forms of chemicals play a role in leaching chemicals out. Total chemical analysis is able to quantify all chemicals in the samples, in spite of their forms of existence. Results of leaching chemical analyses, however, heavily depend on the existence forms of chemicals or chemical distribution among the four phases. Higher fractions of solid, aqueous, and adsorbed phases and lower fraction of crystallized phase lead to higher chemical concentrations in the leachates, and *vice versa*. Acid fluids tend to enhance extractions of adsorbed and solid phases. Larger specific surface of constituents helps a more extensive contact between extraction fluid and constituents, and may result in higher extractability as well. Therefore, to characterize the leaching behavior of foundry sand is complex. To correlate results of total and leaching chemical analyses of foundry sand needs to consider the existence forms of chemicals, constituent gradations, and surface textures.

3.6. *Censored Data*

More often than not, foundry sand characterization data consist of a mix of results that can and cannot be quantified precisely. The portion of nonquantifiable observations, reported as below limits or “not detected,” is “censored data.”

In chemical analyses, the instrument detector provides the electronic signals to register the mass to determine the concentration. The lower the concentration is, the more difficult the signals can be clearly distinguished from other electronic background noise. To be an extreme case, the instrument responses readouts for blank samples, but is not confident about their existence. Therefore, conceptually, there has to be some limit, below which it is not possible to determine if there is detection or not. In that concept, some numerical standard is brought in to specify the limitation of the instrument.

One important standard is called the “method detection limit” (MDL), which is approved by the US EPA. MDL is defined as the minimum concentration of a substance that can be measured and reported with 99% confidence that the analyte concentration is greater than zero and is determined from analysis of a sample in a given matrix containing the analyte.[19] The other standards of detection limits include “quantity detection limit” (QDL) and “low detection limit” (LDL). All number dealing with detection standards lead to censored data.

Frequently censored characterization data impede the application of conventional statistical procedures to obtain summary statistics. Figure 7 illustrates the total cadmium in a set of foundry sand samples, where quantified/detected and censored data co-exist. What interests the researchers, the public, and the regulators is the statistical summaries (e.g., estimations of mean, median, and percentiles) and the data distribution (e.g., normal, log-normal, or Weibull), which is not approachable as the data available is incomplete. Censored observations represent ranges rather than a number. Censoring may range from 0 to 1 depending upon the composition of samples and quantitative level of instruments. It is not easy to simply run statistical programs on data containing both quantified and nonquantifiable data, and is not suitable to substitute arbitrarily for nonquantifiable data. Multiple censoring limits (e.g., <0.1, <0.5, and <0.3) in a data group further complicate the issue.

A number of methods were proposed to address censored data, which are largely classified into three types: substitution methods, parametric methods, and

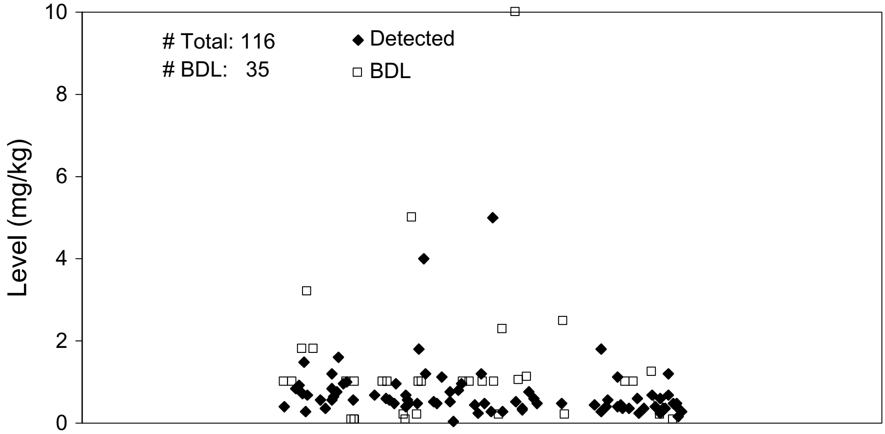

Figure 7. Co-existence of Quantified and Censored Characterization Data.[41]

nonparametric methods. Substitutions represent the conversions of censored data into numbers. The incomplete original data are then converted into complete ones. Common statistical methods can then be applied to obtain statistical implications. Parametric methods include two approaches: regression on order statistics (ROS) and maximum likelihood estimation (MLE). Their validity depends on the given function of the population distribution of an element's concentration and any hypothesis test or confidence interval estimation is base on the assumption that the population distribution function is known, or known except for some unknown parameters. Nonparametric methods include two types: ranking order and survival analysis. They do not assume a particular population probability distribution and are valid for data from any population with any probability distribution, which can remain unknown.

Substitutions are the most easy-learning methodologies addressing censored data. US EPA has traditionally substituted zero or one-half the reporting limits for censored data and computed the average concentration in the usual way. The assumption of equal precision or zero fill-ins is demonstrably false, but the method is computationally simple and often adequate for most practical purpose if the quantifiable frequency is 80% or more.[20,21] With the advent of convenient software, there appears to be no reason to use simple substitutions.

ROS fits a straightline to the normal scores of the order statistics for the quantified observations, and then to fill in values extrapolated from the straightline for the censored observations. Many modified ROS approaches were developed in documentations.[22–25] MLE uses both quantified and censored observations to compute statistics for the entire data set. It calculates the estimated mean and standard deviation by adjusting downwards the statistics of the uncensored observations as a function of the amount of censoring in the data set. The earliest and still most used method was published in two papers by Cohen[26,27] to calculate the estimated mean and variance.

Ranking order uses ranks that may be considered preferable to the actual data. If the numbers assigned to the observations have no meaning by themselves but attain meaning only in an ordinal comparison with the other observations, the numbers contain no more information than the ranks contain. Survival analysis uses the fact for probability reasoning that if the concentration of a sample is below x, its concentration is also below all limits or values above x. Documents[28,29] present the detailed concepts and practical applications of survival analyses to censored data.

Studies[20,23,25,30] have consistently found the ROS and MLE procedures superior to simple substitution methods. However, both MLE and ROS are not indefectible. They assume that the true concentrations in all samples come from a common population that is log-normally distributed. Violation of this assumption can lead to invalid applications of a statistical technique. The decisions and conclusions derived from incorrectly used statistics can be costly in environmental assessment. In this point, nonparametric methods, which do not count on distribution assumption, offer more reliable alternatives for processing censored data.

3.7. *Chemical Types and Quantification*

Major oxides and trace elements are two groups of chemicals related to the characterization of foundry sand. The former refers to the oxides analyzed by using x-ray fluorescence and is reported at a mass percentage level. American Foundrymen's Society (AFS)[17] conducted an investigation of oxides in foundry sands, with average results presented in Table 10. It is indicated that silica occupies averagely 88% by mass of the foundry sand. The latter, trace elements, is rather loosely used in the scientific literature to designate a number of elements that occur in natural systems in small concentrations. As defined in many dictionaries, trace elements are those chemical elements, especially metals, used by organisms in minute quantities but believed essential to their physiology. However, the term is and has been used to designate elements with no known physiological function which, when present in sufficient concentrations, may be toxic to living systems. The trace elements are also the ones frequently called to characterize foundry sand in terms of its environmental compliance.

Of the many different chemicals in nature, the most concerned trace elements, as related to the chemical characterization of foundry sand, are those that are regulated in Tables 6–8, even though additional chemicals are frequently measured depending upon state or local regulations. They cover a wide spectrum of heavy metals that may come from various scrap feed or alloy additions, common light metals that are also hazardous to human health if only digested or exhaled in enough amount, and organics that are stemmed from the intact and decomposed binders. These chemicals play an important role in environmental compliance of foundry sand. For inorganic

Table 10. Chemical Oxide Compositions of Foundry Sand.[17]

Oxides	Percentage by Mass (%)
SiO_2	87.91
Al_2O_3	4.70
Fe_2O_3	0.94
CaO	0.14
MgO	0.30
SO_3	0.09
Na_2O	0.19
K_2O	0.25
TiO_2	0.15
P_2O_5	0.00
Mn_2O_3	0.02
SrO	0.03
Others	5.28

chemicals in Table 6, these chemicals are also commonly referred to as metals, toxic metals, heavy metals, trace metals, transition metals, or micronutrients. Although As and Se have both metallic and nonmetallic properties, they are also commonly called metals. Unlike organic compounds, the unique characteristic of all metals is that they are not biodegradable.

For the variety of organics in Tables 7 and 8, they may exist in liquid or solid form in foundry sands and may exhibit volatile or semivolatile characteristics. They can commonly be classified into following types: hydrocarbons (e.g. benzene, toluene, ethylbenzene, and xylene), chlorinated hydrocarbons (e.g. tetrachloroethylene and perchloroethylene), polycyclic aromatic hydrocarbons (PAHs) (e.g. naphthalene, phenanthrene, and pyrene), polychlorinated biphenyls (PCBs) (e.g. aroclor), and pesticides (e.g. aldrin and endrin).

Metals and organics enter foundry sand throughout most metal-casting operations shown in Fig. 4. The high-temperature-melt metals may fuse sand grains together or penetrate into their pores or concaved spaces. Shake-out metal chips will be more or less mixed with system sands. Sand collections in the casting finish areas contain metal scraps. Incombustible binder residuals and decomposed organics will remain as residues in sands. Although reclamation processes remove parts of the residues, some chemicals eventually accumulate in foundry sand after a number of operation cycles.

Quantification of chemicals, specifically against the threshold values in Tables 6–8, is the principal work scope of foundry sand characterization. The most concerned quantifications include the summary statistics, i.e., mean, median, maximum, and percentile values. When there are not adequate statistical samples, case studies may also aid implicating the characterization.

Documents[31–33] investigated that foundry sands are not hazardous in term of toxicity regulations. Correlations were researched between field leaching levels and laboratory extraction levels,[34,35] among compounds,[36] with external factors.[37] The extractability of multiple waste streams was compared.[34,36] The chemical distributions were discussed in document.[38] Chemical characteristics across 34 foundry sand samples were clustered into four types based upon data correlation.[39] A nice characterization summarization of the above investigations was prepared in a report by Winkler,[40] where observations vary case by case, and only a few projects were completed with a comprehensive statistical analysis that included the determination of a roster of statistical parameters to characterize a vast pool of analytical data.

According to Winkler's collective report,[40] foundry sands largely extract regulated metals well below the toxicity characteristic levels, except for the case of a brass foundry whose foundry sand has an extreme high level of Pb. In addition, one important inference drawn from the correlation of field and laboratory results is that even when the presence of the constituents in a total waste stream can be traced, it does not always imply that they could readily be leached. This is usually because the constituents are either present in forms that are not soluble (e.g., crystallized phase) or are unavailable to the leaching media, which is briefed in section of 3.5. A comparison to a mixed-waste municipal landfill indicates that the foundry sand leaches one to two orders of magnitude less than the typical waste in a municipal landfill.

Adequate statistical samples enable the use of a statistical methodology and offer reliable statistical summary. Based on over 1000 data sets, Deng[29,41] applied survival analyses to obtain the estimates of mean, median, and 95th percentile on the characterization of foundry byproduct, specifically foundry sand, slag, and dust. The effects of factors (i.e., cast metals, leaching protocols, and waste streams), were also analyzed. The data distributions were discussed to fit parametric models. This kind of quantitative characterization helps provide insight into the data and makes more sense of them. Partial results indicate that the 95th percentiles of metallic elements in foundry sand leachates by TCLP, SPLP, and ASTM D3987 are well below the TCLP threshold values. Foundry sands are largely not hazardous. TCLP is more aggressive on foundry sand than SPLP and ASTM D3987. Foundry sand does not pose a greater threat to the environment than soil. The concentrations of most regulated metallic elements are less than or in the same level as those of soil. Foundry sands contain more metallic elements, although not regulated, that are frequently and heavily used in casting process than soil, such ash aluminum, copper, and iron.

Many case studies were also performed to characterize both laboratory and field leaching properties of foundry sand. Leachate studies[42] were conducted, based on materials and waste balance studies, to examine the characteristics of any leachates that can be expected from foundry waste materials. Leachates from ferrous foundries caused less inhibition of light production by the Microtox$^{\text{TM}}$ bacteria than did virgin sands,[43] which means that no real differences were seen between system sands and fresh or aged waste sands. Foundry sands do have potential for highway construction and do not pose serious threat to underground water.[44,45] A metallic iron presence in the foundry sand waste may inhibit lead leaching in TCLP tests.[46]

Two test sites, each using both foundry sands and natural soils for highway embankments, were constructed and monitored over a two-year period.[47] Field lysimeter data indicated low release of parameters tested, most at levels below drinking water standards. Notable exceptions include arsenic, manganese, and total dissolved solid. However, at each site, all three of these parameters were found in similar concentrations and at significantly higher mass quantities in leachates from natural soils. No significant changes in groundwater quality attributable to the foundry wastes were observed.

Aforementioned numerical case studies have verified the nonhazardous attributes of foundry sands. There is a favorable opportunity for foundry sand reuse, instead of to be landfilled, as most of the other marketable byproducts, e.g., fly ash, silica fume, ground blast furnace slag, shredded paper, and alumina cans. A critical step in the reusable program of foundry sand is to seek its suitable applications, verify its performance, and develop application specifications or guidelines.

4. Benefical Reuse of Foundry Sand

Foundry sand, as an aggregate mineral consisting mainly of SiO_2, is originally used as refractory materials for molding and core making purposes in the metal casting workshops. After disposed as one of byproducts from foundry facilities, foundry sand aggregates still exhibit valuable physical, mechanical, and chemical properties, which encourage its reuse potentials in various applications. To better recognize its beneficial reuse, an efficient way is to match foundry sand's properties against those of current in-use products. Foundry sand may coincide with these products in some aspect and secure the opportunity of a full or partial substitute. Pilot- and full-scale laboratory experiments or field case studies should generally be implemented to validate the substitution.

4.1. *Market Opportunities*

The market opportunities of foundry sand are closely associated with its physical, chemical, and mechanical properties. The physical properties mainly include density, specific gravity, gradation, AFS grain fines number (GFN), fines content, particle shape and surface texture, compaction, moisture content and hydraulic conductivity, etc. The chemical properties are related to the major oxide, organic contents, and trace chemical levels in foundry sand. The mechanical properties basically mean the shear strength of foundry sand, as it is a discrete material similar to soils. The shear strength is determined by its cohesion, internal friction angle, and shear behavior.

4.1.1. *Foundry Sand Physical Properties*

Table 11 presents the physical properties of 17 representative foundry sands in an investigation.[41] Bulk density is necessary for use in selecting proportions of mixture, and it is used for determining mass/volume relationship for conversions in purchase agreements. Void represents the space between particles in foundry sand mass not occupied by solid particles. It estimates the space that should be filled by other materials when mixing together. Absorption values are used to calculate change in the weight of an aggregate due to water absorbed in the pore spaces within the constituent particles, compared to the dry condition, when it is deemed that the aggregate has been in contact with water long enough to satisfy most of the absorption potential. Specific gravity is an aggregate characteristic generally used for calculation of the volume occupied by the foundry sand aggregate in any product that is proportioned or analyzed on an absolute volume basis. It is also used in the computation of voids in aggregate and the determination of moisture in aggregate by displacement in water.

Data variations, mostly within acceptable ranges, are found among foundry sands. The variation in specific gravity (2.38–2.72) of foundry sands has been attributed to the variability in fines and additive contents in different samples. Reported values of absorption are found to vary widely, which can be attributed to the presence of binders and additives.[45] Most bonding systems need water as solvent or catalyst to activate various inorganic and organic binder raw materials. Although the melting temperature is high, it decreases significantly along the dimensions of molding. After the refractory processes, low water contents (0–4.85%) are still retained in the foundry sand. About less than 1.35% by mass foundry sand passes 0.075-mm openings, except for the sand from a copper-based facility (9.2%) that uses finer aggregate for finer finishing.

Fineness number (AFS GFN), a measure of average grain size, is based on the number of openings per inch of a sieve that would just pass the average size calculated from the sieving analysis. According to Table 11, steel castings typically use a very coarse sand grain size with a low grain fineness number, such as 45–55 AFS GFN, because they have a lot of gas to get rid of in a hurry. Iron castings use a little bit of finer sand, 55–70 AFS GFN. Nonferrous castings are even finer, 65–90 AFS GFN, which enhances the casting surface finish of nonferrous products.

Table 11. Physical Properties of Foundry Sands.[41]

Sample ID	Metals Cast	Loose Bulk Density (lb/yd^3 (kg/m^3))	Specific Gravity	Absorption (%)	Void of Loose Sand as Oven-dry (%)	Finer than 75-μm by Washing (%)	AFS GFN	Moisture Content (%)
		ASTM C128				ASTM C117	AFS 1106	ASTM C566
FS01	Iron	2457 (1458)	2.503	3.029	48.39	0.482	50.19	1.73
FS02	Iron	2808 (1666.4)	2.722	0.381	49.13	0.078	49.45	0.29
FS03	Iron	2330 (1383)	2.425	2.312	46.91	0.172	51.69	3.17
FS04	Iron	2472 (1466.9)	2.376	2.944	39.99	1.186	76.92	1.42
FS05	Iron	2384 (1415)	2.408	1.688	43.01	0.493	61.26	1.96
FS06	Iron	2214 (1314)	2.546	1.807	56.90	0.000	41.97	1.70
FS07	Iron	2558 (1518)	2.547	1.564	48.73	0.655	60.28	0.00
FS08	Iron	2322 (1378.1)	2.385	3.348	45.86	0.502	56.28	2.03
FS09	Iron	2558 (1518)	2.588	1.092	44.66	0.522	59.18	1.62
FS10	Aluminum	2512 (1491)	2.632	0.523	44.87	1.345	56.68	3.50
FS11	Aluminum	2706 (1606)	2.594	0.860	48.68	0.077	46.92	4.08
FS12	Aluminum	2780 (1650)	2.693	0.787	44.01	0.369	69.99	0.64
FS13	Iron	2662 (1580)	2.638	0.718	42.19	0.515	51.62	0.00
FS14	Aluminum/ bronze	2602 (1544)	2.480	4.148	43.86	9.209	90.36	0.14
FS15	Iron	1859 (1103)	2.379	3.834	55.68	0.414	45.66	4.08
FS16	Steel	2490 (1478)	2.523	3.093	53.83	0.726	54.71	1.02
FS17	Steel	2649 (1572)	2.636	1.171	40.94	0.928	48.68	4.85
Min.		1859 (1103)	2.38	0.38	39.99	0.0000	42.00	0.00
Max.		2807 (1666)	2.72	4.15	56.90	9.2088	90.00	4.85

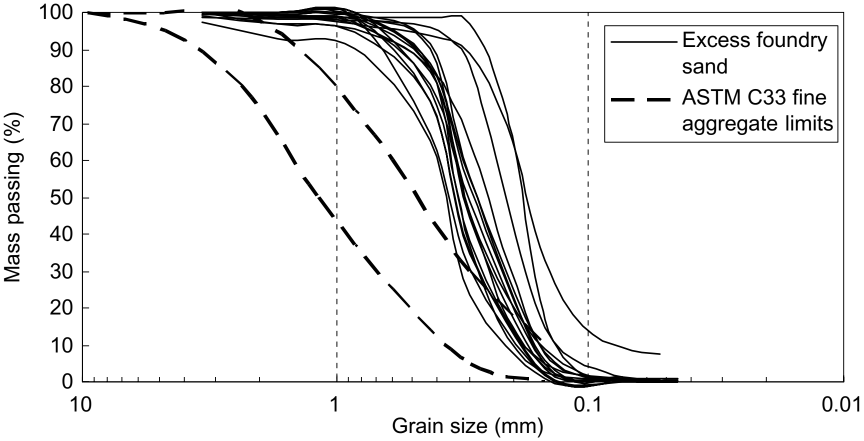

Figure 8. Gradation Curves of Foundry Sands.[41]

Gradation effectively influences the flowable consistency, compaction, and strength of the materials. The gradation curves in Fig. 8 show distributions of particle sizes. It is indicated that gradation of foundry sand is dominated by a common shape. Although grain sizes span from 3.5 mm down to zero, the size is highly concentrated, with 90% in narrow range of 0.15 mm (5% quantile) and 0.80 mm (95% quantile). In addition, within this range, particles are distributed linearly along size. The rest 10% particles are extended two-way at tails. The grain size distribution of foundry sand is more uniform and finer than conventional fine sand.

Four different basic shapes are recognizable for foundry sand: round, sub-angular, angular, and compound or composite. Grain shape is important with respect to flowability and strength. Round sands give superior flowability to angular sands. In addition, angular sands yield higher strength than round sands do. However, round grain has the lowest surface area and requires the least amount of cementitious material. Most of lake sand is in this shape. Angular sands have grains with edges that form acute angles (less than 90°). They are poor for making a good working consistency, for angular particles lockup and neither slide nor compact as well as round ones do. They are made from crushed rock. Compound sands have grains that are stacked together. They look like popcorn balls and do not flow well. When they do break apart, there are a lot of fines.

In size ranges of 0.15–0.80 mm that cover 90% foundry sands, subangular and round grains occupy more than 50%.[41] It was also discovered that as the size increases, the shape tends to be compound or round. If the size decreases, the shape tends to be angular or jagged. It was observed that the compound grains were composed of subangular and round grains once they were crushed under the microscopes. Therefore, subangular and round dominate the shape of sands.

4.1.2. *Foundry Sand Chemical Properties*

The chemical properties of foundry sands are mostly addressed in Sec. 3. It is summarized that foundry sand is silica aggregate containing a small quantities of metal oxides and adsorbed with a variety of trace metals and organic compounds. Quantities of metal oxides can be referred in Table 10. Although foundry sand is, mostly, a nonhazardous material, cast metal type plays an important role in affecting the total chemicals in foundry sand. Copper-based foundry sands tend to leach more copper and lead.[41] Leaching protocol, in conjunction with existence forms of chemicals, also influences metals extractability. In these situations, it is advisable to take into account the variation of foundry sand chemical properties and make full use of individual characteristics in their reuse programs. Analytical chemistry tests are anticipated to characterize the existence forms of chemicals, which determine the mobility and accessibility of chemicals in some reuse applications.

In addition to the chemical contents, chemical activities of foundry sand should also be considered in the reuse programs. These chemical activities include pH value, corrosivity, reactivity, ignitibility, resistivity, etc. Depending on the binder and type of metal cast, the pH of foundry sand varies from approximately 4 to 8.[48] Based on 79 observations of various source foundry sands, Deng[41] found that the pH values of as-received sands largely range from 4 to 10, on average 8.3. It was reported that some spent foundry sands can be corrosive to metals.[49] However, flowable fill containing foundry sand is noncorrosive in nature because of the absence of chlorides and high pH values of 11.4–12.3.[50] The content of organic impurities, particularly from sea coal binder systems, can vary widely and can be quite high. This may preclude foundry sand's use in applications where organic impurities could be important, such as Portland cement concrete aggregate.[51] The loss of ignitition of foundry sand was reported in the range of 0.45–12.1.[17,52] A large fraction of clay lumps and friable particles have been reported, ranging from 1% to 44%, which are attributed to the lumps associated with the molded sand. This fraction can, if required, be easily disintegrated in the test procedures.[17,52]

4.1.3. *Foundry Sand Shear Strength*

Shear strength represents the values of cohesion and internal friction angle, which are often used in geotechnical engineering for deformation and strength calculations. The strength values are closely related to the compaction state which foundry sand belongs to. Dense and compact structure results in high shear strengths and engineered shearing behavior, which is favorable for infrastructures to bear applied loads. Table 12 presents the typical values of shear strengths of foundry sands and construction sands in loose and dense states. Spent foundry sands present lower shear strengths, specifically cohesion, than virgin foundry sand in both loose and dense

Table 12. Shear Strength of Foundry Sands and Construction Sands.[52–55]

	Loose State			Dense State		
	D_r (%)	c' (kPa)	ϕ' (°)	D_r (%)	c' (kPa)	ϕ' (°)
Spent foundry sand	29	4.14	32.4	90	9.93	36.3
Spent foundry sand	34	5.17	34.2	98	12.55	40.9
Spent foundry sand	—	6.8	35.0	—	13.10	38.0
Virgin foundry sand	—	13.8	32.6	—	15.20	37.0
Uniform medium sand	—	—	—	—	0.00	35–38
Uniform sand	—	0.0	29–30	—	0.00	36–41

states. It is interpreted that the major binders in virgin foundry sand are worn out after several casting cycles, and the cohesion is decreased for spent foundry sands. Spent foundry sands show better cohesion than and comparable internal friction angle to construction sands.

Foundry sand has good durability as measured by low Micro-Deval abrasion, <2% loss[49] and magnesium sulfate soundness loss tests, ranging 5–15%.[48] The Micro-Deval abrasion test is an attrition/abrasion test where a sample of the fine aggregate is placed in a stainless steel jar with water and steel bearings and rotated at 100 rpm for 15 min. Recent study[52] reported relatively high soundness loss, ranging 6–47%, which is attributed to samples of bound sand loss and not a breakdown of individual sand particles.

4.1.4. *Foundry Sand Reuse Markets*

Foundry sands are essentially discrete fine mineral grains encompassing a minimum amount of clays, resin/oil binder, and additives, and a trace amount of metals and organics. From a technical standpoint, foundry sand has engineering and agricultural properties that make it a good fit for many applications. According to its properties, foundry sand is acknowledged usable in most of the applications where fine mineral grains (e.g. conventional construction sands, soils) are used. Abichou developed a database[56] to manage the foundry sand reuse projects in North America. In this database, 98 nationwide projects were categorized into applications of flowable fill, embankment and subgrade, Portland cement concrete, asphalt concrete, Portland cement manufacturing, landfill cover and linear system, and soil amendments. In terms of disciplines that end-use products belong to, these reuse applications cover civil, transportation, material, and agricultural engineering. In terms of the reuse manners, foundry sand can be reused either alone or conjunctly with other materials. Table 13 presents the foundry sand reuse programs. Engineering-related applications (e.g. civil, transportation, and materials) mainly make use of physical and mechanical

Table 13. Foundry Sand Reuse Programs.

					Main Reactions	
Applications	Disciplines	Manner	Other Components	Involved Properties	In Production	In Use
Barrier, liner, cover	Civil	Single	Sometimes clay	Compaction, fine content, hydraulic conductivity, gradation	Physical	Physical, chemical, few biochemical
Highway embankment	Transportation	Single	—	Compaction, shear strength	Physical	Physical
Flowable fill	Civil	Conjunct	Cement, fly ash, water	Gradation, particle shape, fines content	Physical, chemical	Physical
Portland cement	Materials	Conjunct	Limestone, shale, etc.	Contents of SiO_2, Al_2O_3, and Fe_2O_3	Physical, chemical	Physical, chemical
Cement concrete	Civil/materials	Conjunct	Cement, coarse, and fine aggregates, water	Gradation, clay content, oxide contents	Physical, chemical	Physical, chemical
Asphalt concrete	Civil/materials	Conjunct	Bituminous asphalt cement, coarse, and fine aggregates	Gradation, shape, clay content	Physical, chemical	Physical
Soil amendments	Agricultural	Conjunct	Compost, nutrients	Metal oxides, organics, trace chemicals	Chemical, biochemical	Physical, chemical, biochemical

properties of foundry sands. Agricultural applications generally rely on the chemical properties of foundry sand.

Physical changes and/or chemical and biochemical reactions occur, specifically on foundry sand, in the production and in the use of end-use products. Physical changes (e.g. compaction and gradation) happen for engineering-related applications, such as hydraulic barriers, landfill compact liners and cap covers, and highway embankments. Chemical and biochemical reactions occur in agricultural applications (growing mixes and topsoils). Cement, concrete, and flowable fill productions undergo both physical changes and chemical reactions. There are also physical and/or chemical variations related to the end use of these products. For examples, landfill liners not only subject to the seepage of leachates, but also are exposed to the erosion of chemicals in the leachates. Reactions involving multiple phases (e.g. micrograms, nutrients, and foundry sands) exist in the matrix of topsoil.

Foundry sand aggregates experience different phase transitions in the productions of these products. Aggregates are fully or partially converted into new matters in cement, concrete, flowable fill through chemical reactions. Foundry sand aggregates basically remain unchangeable in growing mix and topsoil, where the adsorbed chemicals in foundry sand participate the chemical reactions and are converted into new matters.

4.2. *Hydraulic Barrier, Landfill Liner and Cap Cover*

4.2.1. *Description*

Hydraulic barriers, landfill liners and cap covers are low-hydraulic-conductivity geotechnical works used mainly for contaminant containment and low precipitation penetration purposes in the subsurface or landfills. Barrier, or termed as cut-off wall, is constructed in the subsurface to mitigate contaminant transport by minimizing the movement of groundwater from a contaminated site to uncontaminated areas or by preventing clean groundwater from entering a contaminated area. Barrier is generally constructed vertically as a relatively impermeable wall that either contains or redirects groundwater flow. Liner and cover are two important parts in a landfill. Liner is constructed at the bottom of a landfill to isolate the landfill leachate from migration, similar to the role of the barrier. Cover or cap is set on the top of a landfill to act as a barrier layer blocking or minimizing the infiltration into buried waste and facilitating the runoff.

Barrier, liner, and cover commonly consist of compacted clay, soil-bentonite, and cement-bentonite mixtures. They are normally compacted or rolled in lifts, or grouted as a slurry into the subsurface to provide a low hydraulic conductivity of 1×10^{-7} cm/s or less. The low hydraulic conductivity criterion is effective in both limiting the water seepage volume (advection) and impeding the dissolved chemical transportation (diffusion). The low hydraulic conductivity property is commonly gained by compacting fine well-graded particles (e.g. clay), or using self-compaction/set materials (e.g. grouted cement-bentonite mixture).

4.2.2. *Foundry Sand Opportunity*

Green sand is the most used type of foundry sands and occupies the majority of disposed foundry sands. Its primary components are fine silica sand, up to 10% bentonite and water. The gradation ranges of green sand are close to those of natural sandy and silty soils. The sand-bentonite structure of green sand is similar to the soil-bentonite mixture of a barrier. This property enables green sand to be a potential substitute for compacted clay or soil-bentonite mixture in the barrier, liner, and cover. Chemically bonded foundry sand can also be amended to present a compact structure and low hydraulic conductivity by incorporating a proper amount of clay contents.

4.2.3. *Research Experiences*

Low hydraulic conductivity compliance is usually of paramount importance in qualifying foundry sand reuse in barrier, liner, and cover applications. Clay content and compaction effort are two major factors affecting the hydraulic conductivity of foundry sand. Research[57] at the University of Wisconsin-Madison shows that foundry sands having liquid limit above 20%, plasticity index above 3, or bentonite content above 6% can be compacted to achieve hydraulic conductivities below 1×10^{-7} cm/s. The hydraulic conductivity of foundry sands is not as sensitive to compaction water content and compaction energy as the hydraulic conductivity of compacted clays. Similar results were reported for compacted sand-bentonite mixtures in the same investigation. Consequently, it may be easier to construct barrier layers with foundry sands than with clays.

Hydraulic conductivity testing was also performed on selected foundry sands using a 0.1-N $CaCl_2$ solution, a 0.05-N acetic acid solution, deionized water, and a leachate from a local municipal solid waste landfill.[57] Results of the chemical permeation tests show that type of permeant has little effect on the hydraulic conductivity of foundry sands, except for the acid solution.

Barrier layers constructed with foundry sands are expected to be unaffected by freeze-thaw and desiccation.[57] Therefore, barrier layers constructed with foundry sands will perform better than similar layers constructed with clayey soils. However, the resistance to freeze-thaw and desiccation should be evaluated on a case by case basis.

Kunes and Smith[58] concluded that compacted foundry sands have hydraulic conductivities comparable to those of clayey materials typically used in the construction of cover systems. During their study, selected foundry wastes were also successfully mixed with soil and used as a substitute for the vegetative layer in landfill covers.

Construction of barrier, liner, and cover using foundry sand should be performed according to the same specifications using conventional materials. Before any design, the relationships between compaction water content and effort should be determined in the laboratory. These relationships can then be used to design acceptable field compaction criteria.

4.3. *Highway Embankment*

4.3.1. *Description*

Highway embankment is a rigid standing on the ground to carry roadways and support the traffics. Highway embankment earthworks are generally accomplished by filling soils compacted in lifts. The fills need to be reinforced adequately to eliminate any directional deformation or swelling under working conditions.

Technically, highway embankment fills should be compacted to a minimum unit weight of 2,565 lb/yd^3 (1,522 kg/m^3) and at a water content around optimum water content. Most specifications require a maximum liquid limit of 65%, and a plasticity index less than the liquid limit minus 30.

4.3.2. *Foundry Sand Opportunity*

Highway embankment fill materials can be the market of excess foundry sand. Most foundry sands present volumetric stability only if the bentonite contents in green sands are low and inactive to avoid any swelling. Physical index tests arguably suggest that foundry sand can be densified with appropriate water contents and compaction efforts. Therefore, foundry sands are eligible to be considered as embankment fill materials. Since the embankment fill volume is relatively vast, a sufficient supply of qualified foundry sand is generally needed to complete a section of embankment earthworks.

4.3.3. *Research Experiences*

Compaction factors and effects are mostly addressed in researches with regard to the applications of foundry sand as highway embankment fill materials. Javed[45] and Mast[53] concluded that it is generally advantageous to compact green sands at a molding water content range of −3–0% of standard effort optimum moisture content, i.e., moisture contents of 12–15% compacted by using a rubber tire roller with a minimum weight of 356 kN with a 30 psi (207 kPa) tire pressure in their cases. The resultant typical resilient moduli were equal to or greater than moduli of state regulated subgrade soils. The geotechnical investigation[59] conducted at

Purdue University monitored the deformation of the embankment constructed with foundry sand using settlement plates and inclinometers. The project demonstrates that foundry sand can perform as well as a structural fill, having strength and deformation characteristics comparable to natural sand.

Field project by Mast[53] also demonstrates a positive performance of foundry sand in highway embankment applications, where three kinds of fill materials were compared regarding their deformation in one-third section of the embankment. The fill materials include construction sand (control section), clay (control section), and green sand (test section). It is suggested that green sand section performs as well as the other control sections.

Users should determine the grain size distribution, Atterberg limits, bentonite content, and the moisture–density relationships of foundry sand prior to its use in highway embankments. Hydraulic conductivity and swell potential are two major geohazard triggering factors in the highway embankments. In the case of foundry sands, these two factor properties are reliant on active clay content and amount of material smaller than 200 mesh.[60]

4.4. *Flowable Fill*

4.4.1. *Description*

Flowable fill, also termed as controlled low-strength material (CLSM), is a self-compacted, self-set, marginally cementitious geomaterial used primarily as a backfill in lieu of compacted backfill. It has the following features listed by American Concrete Institute (ACI) Committee 229[61]: (a) it is a self-leveling, liquid-like material, and self-compacting to 95–100% of the maximum unit weight with minimal effort and no vibration or tamping and (b) Its maximum unconfined compressive strength (UC) at day 28 is 1,200 psi (8 MPa). Most flowable fill applications require unconfined compressive strengths of 200 psi (1.4 MPa) or less to allow for future excavation using hand tools or light digging equipment.

Flowable fill contains necessarily cement and water. It may also contain fly ash, fine aggregates, or chemical admixtures in proportions such that the final product meets the strength and flow consistency requirements. The components are prepared and blended into a slurry state that is checked by running a flow consistency test. With the addition of water, the cement hydration process proceeds in the mixture. Resultant hydrates offer certain amount of binding force and enable the set of mixtures. The hardened mixture thus behavior like a strengthened geomaterial and is able to bear loads.

Flowable fill is used widely as backfills for lots of infrastructure works (e.g., utility cuts, trenches, pipes, vaults, abandoned underground pipes, culverts, tanks, mines, sewers, voids under roadways, foundations, and retaining walls), load supporters for a variety of bearing structures (e.g., pipe bedding, road base, bridge approaches, paving sub-base, floor slab sub-base, liner base, cover mats, and replacement of poor quality soil). The former (backfills) is generally called excavatable flowable fill with unconfined compressive (UC) strength less than 200 psi (1.4 MPa) to allow a future manual excavation or dig using light equipments. The latter (load supporters) is structural flowable fill with UC strength between 200 and 1,200 psi (between 1.4 and 8 MPa) without excavation demands or dug out by using heavy equipments. The main construction advantages of flowable fill over conventional backfills can be summarized as: limited labor, equipment and inspection, accelerated construction, ready placement at inaccessible locations and improved safety, improved working consistency, improved strength control, limited subsidence and volume instability, uniform density and load, low hydraulic conductivity, and easy excavation in the future.

4.4.2. *Foundry Sand Opportunity*

Flowable fill is a growing market that meets the basic technical qualifications to address a beneficial reuse program of foundry sand. Flowable fill is typically a mixture of fine aggregate (e.g. gravel sand), fly ash, cement, and water. Since fine aggregate is the major component of flowable fill, there is an attraction of replacing the fine aggregate with foundry sand.

Foundry sand is basically a fine mineral aggregate. According to investigation,[41] more than 90% by weight of the grains falls between 0.15 and 0.80 mm, much finer than the comparable ranges of 0.30–4.75 mm for fine aggregate. The AFS GFN, a foundry industry term indicating the average grain size (the bigger the GFN, the finer the average grains), ranges between 40 and 90 for general foundry sands, higher than around 40 for fine aggregate. Both gradation and AFS GFN suggest that foundry sand is finer than fine aggregate, which aids the flow and compaction of flowable fill. In addition, foundry sand is a uniform equidimentional round-subangular aggregate according to the microscopic observations. The shape characteristics of foundry sand also enhance the flow consistency of flowable fill when foundry sand is incorporated into flowable fill mixtures.

Substitution can be validated by conducting pilot- and full-scale laboratory experiments, and field constructions and investigations. The main topics concentrate on the percentage of substituting foundry sand for fly ash or fine aggregate, the technical performance and environmental impact of foundry sand flowable fill, and the developments of specifications and construction guidelines. Researches need use foundry sands, cement, fly ash, and water as the raw feed. The critical performance parameters of flowable fill include its flow consistency, UC strength gains, hydraulic conductivity, and environmental impacts.

4.4.3. *Research Experiences*

Javed and Lovell[62] prepared flowable fill mixtures using Class F fly ash, Type I cement, and a foundry sand. Mixture designs were tested for flow consistency, set time, and UC strength. It was concluded that foundry sand mixture perform better than conventional flowable fill mixture. Bhat and Lovell[50,63,64] conducted an extensive investigation regarding the economy of foundry sand reuse, flow characteristics, hardening patterns, 28-day compressive strength, long-term strength, pore size distribution, hydraulic conductivity, pH of the pore water, and stress–strain characteristics. It was concluded that flowable fill, containing up to 55.5% foundry sand, is an economic alternative to conventional compacted fills. A systematic procedure for flowable mix design was developed. Works[65,66] also indicated that the flowable fill containing Class F fly ash and foundry sand from ferrous castings are environmental benign.

Naik also performed investigations[65,66] in reusing foundry sand to flowable fill. Document[65] pointed out that foundry sand can be used in flowable fills to replace up to 85% of the fly ash used in the reference mixture, and that the hydraulic conductivity of the flowable mixtures was affected by an increase in either the water to cementitious materials ratio or the foundry sand content. Document[66] used three green sands from ferrous foundries and two Class F fly ashes to product flowable fills. It was presented that a minimum hydraulic conductivity occurs when 30% of the fly ash is replaced with foundry sand. Flowable fills, having 70% of the fly ash replaced with foundry sand, do not have significantly different hydraulic conductivities. When 85% fly ash is replaced with foundry sand, the hydraulic conductivity increases dramatically. Low hydraulic conductivity is also correlated to water to cementitious material ratios ranging from 0.4 to 0.6. The pH of pore solution of hardened flowable fill indicated that the potential for corrosivity was low.

It was reported that flowable fill matrix can stabilize both metals and organic chemicals.[41] For the bleed water and TCLP leachate from foundry sand flowable fills, all regulated metal and organic concentrations, both quantified and unquantifiable, are below US EPA TCLP toxicity criteria and well less than their contents in as-received foundry sands.

Management studies were conducted to enhance foundry sand flowable fill applications. Case studies regarding foundry sand flowable fills were managed in the form of a database.[56] A method for determining the cost of flowable fill and how it can affect a contractor's total construction costs was described.[67]

4.4.4. *Specifications*

To document the specifications of flowable fill containing foundry sand puts it competitively advantageous against other marketable solid wastes (e.g. fly ash and slag) and other foundry sand's reuse programs (e.g. concrete and topsoil). Currently, the use of flowable fill on construction projects is gaining popularity, and foundry sand is increasingly approved by some state transportation departments as a construction material in flowable fill. Many state specifications have been developed or are being developed to standardize the formulations and uses of flowable fill.

US Pennsylvania Department of Transportation (PennDOT) has developed full specification[68] for flowable backfills. The specification covers the material selections and formulation, and the construction guidelines. Major materials include cement, pozzolans, fine aggregate, and water, where foundry sand is included as one of fine aggregates and meets US PennDOT fine aggregate specification.[69] Four types of flowable backfills (Types A to D) are specified in terms of the materials and suitable end applications. Types B and C need to include fine aggregate (e.g. foundry sand). Their mix formulations are presented in Table 14, where units are used based on the unit volume yield of fresh fills. Water volume is not specified but may be added to maintain flow consistency as needed. Type B flowable backfills are made to meet the need of future excavation of the backfill, which is generally applied to utility trenches, pipe trenches, bridge abutments, and around box or arch culverts. Type C backfill will not be excavated and is used to replace unsuitable soils below structure foundations, filling abandoned conduits, tunnels and mines, and backfilling around pipe culverts where extra strength is required.

The flowable fill specification[70] developed by the US State of Iowa Department of Transportation (Iowa DOT) allows the use of fine aggregate, fly ash, cement, and water. Fine aggregate is defined as natural sand consisting of mineral aggregate particles or foundry sand from the castings of ferrous material, and should satisfy the grading requirements in the specification. The basic proportioning for flowable fill mortar is presented in Table 15. Mixes utilizing foundry sand may require more water. The quantity of water used for the trial mix or at the project may require adjustment to achieve proper solids suspension and optimum flow ability.

The specification[71] of US Ohio Department of Transportation (ODOT) refers flowable fill as low strength mortar backfills. The fills mainly contain cement, fly

Table 14. US PennDOT Mix Design and Strength Criteria of Flowable Backfill.[69]

Materials and Criteria	Type B	Type C
Cement (lb/yd^3 (kg/m^3))	50 (30)	150–200 (89–120)
Pozzolans* (lb/yd^3 (kg/m^3))	300 (178)	300 (178)
Bottom ash or coarse aggregate or fine aggregate** (lb/yd^3 (kg/m^3))	2600 (1543)	2600 (1543)
Compressive strength, day 28 (psi (MPa))	125 (0.86), max	800 (5.51), min

*Fly ash, ground granulated blast furnace slag; **natural sand, manufactured sand, or foundry sand.

Table 15. US Iowa DOT Mix Design of Flowable Backfill.[70]

Materials	Quantities
Type I cement (lb/yd^3 (kg/m^3))	100 (60), max
Fly ash (lb/yd^3 (kg/m^3))	300 (178)
Fine aggregate (lb/yd^3 (kg/m^3))	2600 (1543)
Water (gal/m^3 (L/m^3))	70 (345)

Table 16. US ODOT Mix Design of Flowable Backfill.[71]

Materials	Type 1	Type 2	Type 3
Cement (lb/yd^3 (kg/m^3))	50 (30)	100 (59)	0 (0)
Fly ash, Class F (lb/yd^3 (kg/m^3))	250 (148)	*	1500 (890)
Fly ash, Class C (lb/yd^3 (kg/m^3))	0 (0)	0 (0)	500 (297)
Sand (lb/yd^3 (kg/m^3))	2910 (1727)	2420 (1436)	0 (0)
Water (Target) (lb/yd^3 (kg/m^3))	500 (297)	210–300 (125–178)	850 (504)

ash, sand, and water. Foundry sand is mentioned as an alternate material of sand in the mix. Fills are divided into three types by adjusting the mix designs (Table 16). All types shall have UC strengths between 50 and 100 psi (0.35 and 0.7 MPa) at day 28. Their long-term (12 month) UC strengths shall be less than 100 psi (0.7 MPa). The specification also provides detailed information about mixing and placement of flowable fills.

Investigation by Deng[41] is dedicated to the specification developments of foundry sand flowable fills taking into account the variation of foundry sand. Cement, fly ash, foundry sand, and water were proportioned to meet requirements of both excavatable and structural flowable fills. It is found that fines and metal oxides content in the foundry sand play a role in affecting the mix proportions and strength gains of flowable fill. It is suggested that a pilot-scale test program is better proposed and conducted to scout the final mix formulation.

4.5. *Portland Cement*

4.5.1. *Description*

Portland cement, a popular hydraulic cement, is a binder, which generally undergoes hydraulic chemical reactions under water contacts, and sets and hardens to bond other materials together. The bonded mixtures are able to retain strength and stability for a long duration. Resultantly, the most important use of Portland cement is the production of mortar and concrete, the bonding of natural or manufactured aggregates to form a strong and durable building material.

Portland cement is a product of a kiln operation, where ground raw materials undergo chemical transformation. Raw materials used in Portland cement manufacturing must contain the appropriate proportion of calcium oxide, silica, alumina, and iron oxide. Most of these necessary ingredients are usually contained in shale, dolomite, and limestone. According to ASTM C150 that standardizes the Portland cement specification, Portland cements need to meet with the composition requirements of silica, alumina, iron oxide, etc. When deficiencies in these ingredients are encountered in the rock feeds normally used in cement production, additional silica, and alumina are added.

4.5.2. *Foundry Sand Opportunity*

Since foundry sand is a good source of silica, alumina, and iron oxides, it was deemed an attractive ingredient as a raw material for use in Portland cement manufacturing. Chemical reports[41] indicated that over 80% silica by weight is contained in the foundry sand. A useful amount of clay content in the green sand is also a good provider of required oxides. The fine gradation of foundry sand can further aid a complete ingredient chemical reaction in the kiln operation.

4.5.3. *Research Experiences*

A study[17] was conducted to investigate the potential of using foundry sand as a raw feed for Portland cement kiln operation. The study consisted of preparing raw cement feeds with and without foundry sand. X-ray fluorescence analysis of the foundry sand showed that the foundry sand consisted of 88% silica, 5% alumina, 1% iron oxide, 0.19% sodium, and 0.25% potassium. The loss on ignition was 5%. Four samples of raw feeds were prepared and proportioned with limestone, clay, sand, iron ore, and magnesium carbonate. Sand in the mixes was incrementally replaced by foundry sand in percentages of 0% (control sample), 4.45%, 8.9%, and 13.36% (complete replacement by foundry sand). All samples were then ground to pass the No. 200 sieve, and introduced into a kiln at a temperature of 2730°F (1500°C). An oxide and free lime analysis performed on the resulting clinkers showed no significant difference between clinkers obtained from the four different mixes. Four samples of cement were then made by grinding the clinkers and mixing them with gypsum. No significant difference was found between the compressive strength and the set time of the four cements.

According to the Portland cement industry, foundry sand can be beneficially reused in the manufacturing of Portland cement when it possesses the following properties: (a) over 80% silica content, (b) low alkali level, and (c) uniform particle size. Most Portland cement plants require that foundry core butts should be ground to a uniform grain size. Foundry sand addition into the raw feed may vary from kiln to kiln depending upon the quality of their local raw materials.[60] Since different waste streams may vary in their chemical compositions, it is better to divide them according to their streams of origin and to provide a relatively "pure" feed.

4.6. *Cement Concrete*

4.6.1. *Description*

Concrete is one popular form of construction materials that commonly consists of approximately 30% sand (fine aggregate), 50% gravel (coarse aggregate), 15% Portland cement, and 55% water. Under cement hydration processes, concrete mixes eventually hardens and bonding together creating a stone-like material. Concrete can be either cast-in-place or pre-cast into concrete products that are widely used to serve the people. It is used to make pavements, building structures, foundations, overpasses, parking structures, brick/block walls, and piles.

4.6.2. *Foundry Sand Opportunity*

The fine aggregate used in the production of concrete is a potential subject replaceable partially with foundry sand. Selection of fine aggregate needs to comply with specification ASTM C33, which specifies the physical properties of fine aggregate. The properties mainly include gradation, soundness, and chemical contents. Although foundry sands are relatively uniform and fine in size compared to fine construction sand (Fig. 8), they can be a partial replacement for fine aggregates when the combined fine aggregates fall within the gradation ranges and comply with the requirements of chemical contents specified in ASTM C33.

4.6.3. *Research Experiences*

Three foundry sands (a shell sand from a gray iron foundry, a green sand from a gray iron foundry, and a green sand from a steel foundry) were used to investigate their suitability of partial replacement for fine concrete aggregates.[17] Around 33% (i.e., 33 kg of 100 kg of fine aggregates) fine aggregate by weight in concrete mixes were replaced by foundry sands to meet fine aggregate gradation requirements in ASTM C33. The 28-day UC strength of the shell sand concrete was 5,200 psi (36.4 MPa), nearly the same as that of control concrete specimens. The concrete made with the gray iron green sand and steel green sand had a 28-day UC strength of 4,000 psi (28 MPa) and 2,600 psi (18.2 MPa), respectively. The decrease in UC strength of concrete specimens made with green sand was associated with the existence of fines and organic materials in green sands. The researchers also observed that green sand mixes need more water to be workable due to the presence of finer materials. This results in a higher water–cement ratio, which causes the UC strength to decrease. The study concluded that green sand mixes need more cement to yield concrete with acceptable UC strengths.

Further study[60] suggested that normal weight concrete was formulated with a 45% replacement of normal fine aggregate by shell sand and a 9% substitution of green sand for normal fine aggregate to meet all strength requirements.

Naik *et al.*[72] found that concrete specimens containing used foundry sand showed about 20–30% lower values of UC strength at day 28 than concrete specimens without used foundry sand. However, concrete specimens containing 25% and 35% clean/new foundry sand gave almost the same UC strength as that of the control mix. They concluded that the lower strength of the foundry sand concrete specimens is caused by the binders retained in foundry sands, such as bentonite. They also noted that foundry sands can easily be used in lower grade concrete products such as bricks, blocks, and in some structural grade concrete if the mix is designed properly. They recommended further studies on the durability of concrete containing foundry sands.

Approximately 100 full-size concrete blocks were produced and tested in compression after curing times of 7, 14, and 28 days.[73] The replacement of fine aggregates with foundry sand ranged from 15% to 20%. All mix designs used a water–cement ratio of 0.5, air content of 6%, and a target slump of 3 in. (7.6 cm). Control mixes were prepared without foundry sand using the same proportion. Concrete cylinders and blocks containing a blend of used and virgin foundry sands showed compressive fracture strengths that were similar to control specimens without foundry sands. Foundry sand concrete blocks showed inferior compressive strengths. Based on visual inspection, the color of the blocks darkened as the quantity of foundry sand increased in the mix. Blocks with 15% replacement showed minimal darkness while blocks with 30% replacement were considerably darker. Blocks with 45% replacement exhibited colors that were considered too dark to be marketable. It was recommended that 15% replacement of fine aggregates with foundry sand is acceptable in cast in-place concrete and in concrete blocks.

4.7. *Hot Mix Asphalt*

4.7.1. *Description*

Hot mix asphalt (HMA), a prevalent type of asphalt concrete, is one of most popular paving materials used for highways and roadways. HMA consists of a combination of coarse aggregate, fine aggregates, and bituminous asphalt cement. The aggregates are coated and bounded together by bituminous asphalt cement to form a solid unity. The most important properties of the aggregate are gradation, shape, and density, which play the greatest role in affecting the properties of HMA.

4.7.2. *Foundry Sand Opportunity*

The use of foundry sands as fine aggregates in HMA is similar to the case of fine aggregates in cement concrete. Foundry sands exhibit most physical and mechanical properties that asphalt sands require. However, they do differentiate in some aspects, specifically particle gradation and shape, which makes foundry sand unable to be a complete substitution for asphalt sands. Opportunities still exist when the mixed aggregates meet with performance requirements of HMA by a partial replacement of foundry sand with asphalt sand.

4.7.3. *Research Experiences*

Javed *et al.*[74] reported an extensive study on the suitability of foundry sand in asphalt concrete. Five sources of foundry sand were used, including green sands, chemically bonded and shell sands. Suitability was evaluated according to the following characteristics: gradation, deleterious materials, clay lumps and friable particles, durability and soundness, particle shape and surface texture, and affinity for asphalt. Although these characteristics mostly failed the criterion in the specifications of fine aggregates, partial substitution is still a possibility. Fine aggregate substitute rates for green sands will average roughly 10–15% in HMA mix designs, but can be as high as 30% if the clay and organics have been removed. Mixes containing 20% foundry sand typically showed increased need for asphalt binder. Sodium-silicate-based foundry sand appears to degrade the HMA performance.

Moisture susceptibility results[75] indicate that both foundry sand HMA and control HMA were affected by moisture damage. Expanded study should be made in this area to understand the role of the foundry sands in moisture damage. Leidel *et al.*[76] recommended that 15% of clean foundry sand can be used as a replacement of fine aggregates in asphalt concrete. Use of higher percentages of foundry sand without adding anti-stripping additives results in pavement deterioration shortly after construction. If methylene blue clay content is over 10 mg/g, use of anti-stripping additives should be explored.

Gradation of foundry sand, clay, and organic contents in foundry sand may be detrimental to aggregate binder adhesion in HMA. Addition of coarser natural sands and cleanliness operations may help eliminate the issues. Due to highly varied sources of foundry sand, however, the performance of foundry sand in HMA also varies. Physical testing of the sand is absolutely necessary prior to the development of any mix design. Methylene blue, microscope observation, and No. 200 sieving are common methods used for determining clay contents, grain shapes, and fine contents. Susceptibility to moisture damage is another issue related to the volumetric stability of foundry sand HMA.

4.8. *Growing Mix*

4.8.1. *Description*

Growing mix, soil amendment, and top soil may be classified into the same soil mixture type for agronomical purposes. They are typically sandy, silty, or clayey soil grains mixed with fertilizers, compost, nutrients, and micrograms and able to maintain sustainable living substances and energies for rooted crops and plants.

4.8.2. *Foundry Sand Opportunity*

Replacing soil grains with foundry sands in agronomic applications represents an emerging market for beneficial reuse of foundry sand. Loose structure of foundry sand is able to provide connected voids and pathways to transport air, water, and substances. The presence of clay in foundry sand is beneficial because clay increases the capacity of soils to retain nutrients and, therefore, reduce the amount of additional nutrient required for plant growth. A limited amount of metallic, nonmetallic, and organic chemicals in foundry sand may also be the sources of nutrients. Additional nutrients can be added into and mixed with foundry sand depending upon the end use applications.

4.8.3. *Research Experiences*

Dunkelberger and Regan[77] investigated the use of foundry sands as growing mix amendments for horticultural production of ornamental flowers and shrubs. This study was to determine if foundry sands could be incorporated into growing mixes for the productions of Orbit Red and Spring Fever White. Different growing mixes were prepared using foundry sand. Percentage of solids, air, water content, and total porosity were determined for each mixture and compared to those of commercially available potting mixes. Plant growth was evaluated for different types of foundry sands and compared to plant growth in commercial growing mixes. Results indicate that Spring Fever White and Orbit Red will grow in potting mixtures containing foundry sands. It was concluded that a mixture, 50% manufactured growing mix and 50% foundry sand, can be successfully used in growth media or in manufacturing of potting soil for the production of nursery stock. Foundry sands should be mixed with coarse amendment (such as compost) to obtain mixtures with sufficient porosity. The pH of the mix should be kept above 7 to mitigate the potential toxic impact of metals in foundry sands on plants.

Specification ASTM E1598 is usually used to assess the toxicity of soil amendments obtained from mixing industrial by-products with conventional materials. This test should be performed on foundry sand-peat mixtures to demonstrate that foundry sand will not limit plant growth.

Topsoil replacements were blended using foundry sand, papermill sludge, and yardwaste compost.[78]

4.9. *Other Applications*

Other applications, where foundry sand may be reused, include drainage layers, clay bricks, smelting flux, and rock wool. These applications may currently be limited to industrial interests or trial cases, lacking research efforts or detailed open reports.[53] Nevertheless, they may still represent promising markets of foundry sand reuse.

4.10. *Reuse Program Perspectives*

In all of the above applications, foundry sand is required to be separated from other foundry by-products and kept as "pure" as possible. Up to 40 waste streams are generated in a foundry facility (see Table 5),[40] e.g., molding sand, core butt, shot blast, baghouse dust, flux slag, etc. Their physical and chemical properties may vary significantly. In addition, foreign debris in a sand supply can cause adverse damage to a reuser's operation and frustration to the materials manager. To keep clear divisions of foundry sands help maintain their uniformity and present qualified performance in reuse programs.

Unanimously, foundry sand should be tested about its gradation, Atterberg limits, clay content, moisture–density relationship, shape, or chemical composition before the determination of its suitability to any reuse program. In addition, the available volume and transportation of the reusable foundry sand should also be assessed from the standpoint of a contactor, as some reuse application (e.g. highway embankment) may consumes large quantities of foundry sand. Another critical concern is the commitment on the part of the foundry sand provider to seek approval from the state and put a determined effort in preparing the paperwork, as the regulations on nonhazardous solid waste reuse may vary state by state.

The economic consideration in foundry sand reuse is actually a not well answered issue. The fact that foundry sand is a valuable byproduct is accepted, but it is not axiomatic that which program explores the most value of foundry sand and offers the most savings. To run a comparison among a variety of reuse programs according to the unique characteristics of individual foundry sand stream from technical, economic, and environmental standpoints may help decision makers sort each stream to a most suitable and profitable reuse program. In addition, the reuse costs must be compared to those of disposal and reclamation associated with foundry sands to determine the most economic solution for a foundry.[79] Almost always, the spent foundry sand must be processed before the reuse. This processing costs money.

Usually, foundries must pay a recipient to cover the cost of processing the spent sand and any necessary changes to the manufacturing process at the recipient's plant. Employing more than one of the above practices (reduction, reclamation, and reuse) will likely be necessary if the cost of waste is to be minimized and the cost of production is to be optimized.

A study in 1993,[80] although relatively aged, may help indicate the appropriate expenses associated with foundry sand reuse. The analysis indicated that spent foundry sands were brought to a centralized facility for a cost of 3.40 USD/t. A relatively simple processing system, consisting of magnetic separation, screening, crushing, and storage, was able to produce a product that could be used for asphaltic pavement aggregate replacement, landfill daily cover, and liner bedding. The total unit cost, including operation, capital, and transportation, was estimated as 9.80 USD/t of processed foundry sand. These costs must either be charged to the generating foundries or be recouped from the sale of product sand.

5. Case Studies

The case studies presented in this section are sourced from short reports posted on Web sites of Foundry Industry Recycling Starts Today (FIRST) and US EPA. They are recently accomplished field works using disposed foundry sand.

5.1. *Foundry Sand as an Asphalt Pavement Ingredient — Indiana*[81]

Weil-McLain is a gray iron foundry in Michigan City, Indiana, USA. In 2003, Weil-McLain partnered with Reith-Riley, a large Indiana paving contractor, to use foundry sand in a commercial asphalt mix. The asphalt was used at the AM General Test Track for Hummers in Mishawaka, Indiana.

Reith-Riley transported the sand, incorporating it into their existing transportation scheme by picking the sand up from Weil-McLain's facility as a backhaul. Weil-McLain compensated Reith-Riley for using the sand, with a portion of the compensation being used to cover Reith-Riley's transportation costs. This was an advantageous arrangement for Weil-McLain, as the foundry paid Reith-Riley only 25% of what the local landfill fee was at the time. In the end, roughly 4000 tons of foundry sand was used in this project. The foundry sand made up roughly 10% (by weight) of the asphalt aggregate. The foundry sand was made up of a variety of types: less than 10% green sand, less than 25% no-bake sand, around 25% cold-box sand, and around 40% warm-box sand, and was tested for several paving specifications, including density (100–110 lb/ft^3 (1600–1763 kg/m^3)), bulk specific gravity (2.61), soundness (6.9%), and uncompacted void content (33.2%).

This project was possible because Indiana State Law accommodates several beneficial uses of foundry sand under its Type III restricted waste classification. Foundries can apply for the classification by testing their sand and submitting a formal application (with testing results) to the Indiana Department of Environmental Management. Pending test results, the agency can issue a classification that is valid for up to five years.

The benefits of this project went primarily to Weil-McLain. Because they paid Reith-Riley only about 25% of what it would have cost to landfill their sand, they saved over 50,000 USD on disposal costs for their foundry sand over the course of the entire project. There were no economic benefits for Reith-Riley, as they operate local aggregate sources near their asphalt plant. However, because Weil-McLain did compensate Reith-Riley for using the foundry sand, it did make the business relationship economically viable. Additionally, Reith-Riley reported that they believed using the foundry sand contributed to produce a smooth-finish surface for the asphalt. Both Reith-Riley and Weil-McLain were pleased with the results of this project, and Weil-McLain intends to seek out other similar projects for its foundry sand in the future.

5.2. *Foundry Sand as Structural Fill and Road Base — Pennsylvania*[82]

Littlestown Foundry, Inc. (Littco) in Littlestown, Pennsylvania, USA, produces commercial aluminum castings and has been in operation since 1916. Since the mid-1990s, Littco has been supplying most of its spent foundry sand, between 300 and 400 tons annually, free of charge to the Borough of Littlestown, Pennsylvania, for use in civil construction projects.

One project is now complete and two more are currently in progress. The completed project is a new street (Bridge Street) and an embankment built over an abandoned railroad bed. Littco purchased the abandoned railroad right of way and sold it to the borough for US$1, allowing the borough to avoid uncertainties in the legal process of purchasing the right of way. Once the purchase was complete, Littco began transporting its spent foundry sand less than 300 feet (100 m) to the construction site and assumed the cost of labor for transporting and applying the material. Littco's "in-kind" cost was about US$20,000, and the foundry received a Beneficial Use Permit from the Pennsylvania Department of Environmental Protection for this project.

The second set of projects undertaken by Littco and the Borough of Littlestown operated the same way and cost about US$10,000 to date, with an ongoing cost of about US$2,000 annually. They involve using Littco's spent sand as a base fill for a truck turn-around and storm sewer. Like the Bridge Street project, both the turn-around and the borough's storm sewer are directly adjacent to Littco's property. In both, the completed project and the ongoing projects, Littco transports and applies its spent foundry sand to those sites.

All of these projects were pursued because they offer economical and practical benefits to both the Borough of Littlestown and Littco. The borough benefits primarily from having a free supply of fill for the projects, while Littco saves approximately US$35 per ton on disposal costs for its spent foundry sand. Furthermore, the projects use 100% of Littco's spent foundry sand, and the extremely close proximity of the sites to Littco's facility makes transportation and application both easy and convenient. Finally, both the borough and Littco benefit from having improved infrastructure at a relatively low cost.

Littlestown Foundry is extremely pleased with all the projects due to the benefits experienced for both Littco and the borough. Though Littco may have to return to more traditional disposal methods for its spent foundry sand once the truck turn-around and storm sewer projects are complete, they hope to be able to find additional beneficial uses in the future that will be both practical and economical for all parties involved — government, end users, and Littco.

5.3. *Foundry Sand as Soil Amendments — Ohio*[83]

For the past 20 years, Kurtz Brothers, Inc. has collected foundry sand from foundries in Ohio for use in a variety of soil-blending products, including compost, topsoil, and mulch. The vast majority of the sand collected from foundries by Kurtz is generated by ferrous operations, with the remainder coming from aluminum casting operations. Of the approximately 20 foundries from which Kurtz hauls sand, the Ford Foundry in Brookpark, Ohio, has been Kurtz's largest and most consistent source.

In 2001, Kurtz collected approximately 270,000 tons of sand: nearly 241,000 tons from the Ford foundry, and a combined total of 28,000 tons from other foundries. During the same year, Kurtz used approximately 238,000 tons of foundry sand in its products, including 77,240 tons in the production of 180,000 yd^3 (138,000 m^3) of soil blends. Kurtz sells its soil products to approximately 225 customers consisting primarily of garden centers/nurseries and landscapers.

Dugan's Garden Center is a typical customer of Kurtz. Dugan's began to carry blended soil products from Kurtz in the late 1980s and currently carries three Kurtz products: Pro-Blend, which is a primary topsoil product consisting of foundry sand and organic materials, UltraCompost, a biosolids-based compost, and another organic, leaf-based compost. In terms of volume, Dugan's purchases and in turn sells 2,200–2,500 yd^3 (1,680–1,910 m^3) of Pro-Blend a year.

By many accounts, Kurtz's Pro-Blend topsoil is superior to other manufactured soils not made with foundry sand. The Pro-Blend topsoil and other Kurtz blended soils containing foundry sand are looser, less compact, more permeable, and easier to work with than the local topsoil, which has a relatively high clay content. In addition, Dugan's Garden Center first chose to carry, and has continued to carry, Kurtz's topsoil product because it has always been price-competitive. Those interviewed for this case study were satisfied with the performance, engineering characteristics, and quality of final soil products containing foundry sand.

6. Summary

Foundry sand is proven to be a largely nonhazardous solid waste. The ferrous-based and alumina-based foundry sands are generally nonhazardous according to the federal relations. Some heavy metals (e.g. Pb and Cu) are sparsely, not frequently, found exceeding toxic threshold values in copper-based foundry sands. Occasionally, the residue of metal chunks may simply violate the toxic criteria and lead to a hazardous judgment of a waste stream. SPLP is a leaching protocol suitable for simulating the reuse programs where only rainfall or groundwater extractions occur. This protocol should be selected as the toxic assessment in the foundry sand reuse applications.

Beneficial reuse of foundry sand is increasing in a variety of areas involving transportation, construction, and agricultural industries. These industry sectors can use large quantities of foundry sands for earth fills, hydraulic control barriers, concrete aggregates, cement raw feeds, and growing amendments. Providing its full or partial replacement of the original materials (e.g. fine aggregate and clay), foundry sand is able to present competitive performance in the end-use products or applications. Among the reuse programs, flowable fill application is well established with construction specification and guideline developed. Other applications are being developed with promising achievements. Many case applications have been accomplished in construction-related fields, e.g., embankment fills, subgrades, and roads.

In all applications, foundry sand is required to be separated from other foundry byproducts (e.g. dust and slag) and foreign debris (e.g. metal chucks or wooden materials). Foundry sand should also be sourced from a singular bonding system (e.g. green sand or shell) to maintain its uniformity and purity. Moreover, tests of index properties, including grain size distribution, Atterberg limits, clay content, and the moisture–density relationship, should be determined before a reuse program can be implemented, as the foundry sand may vary case by case in its physical and chemical properties. Different physical properties result in varied performance of foundry sands in the produced products.

Nomenclature

Symbol	**Definition**
AAS	atomic absorption spectrophotometry
ACI	American concrete institute
AFS	American Foundrymen's Society
ASTM	American Society of Testing Methods
ASTM D3987	shake extraction of solid waste with water
BDL	below detection limit
c'	cohesion, kPa
CFR	code of federal regulations
CLSM	controlled low-strength material
Dr	relative density, %
EPA	Environmental Protection Agency
FIRST	foundry industry recycling starts today
GC	gas chromatography
GC-MS	gas chromatography-mass spectrometry
GFN	grain fines number
HMA	hot mix asphalt
HPLC	high-performance liquid chromatography
ICP-MS	inductively coupled plasma-mass spectrometry
Iowa DOT	Iowa Department of Transportation
LDL	low detection limit
MDL	method detection limit
MLE	maximum likelihood estimation
ODOT	Ohio Department of Transportation
PennDOT	Pennsylvania Department of Transportation
QDL	quantity detection limit
RCRA	resource conservation and recovery act
ROS	regression on order statistics
SPLP	synthetic precipitation leaching procedure
TCLP	toxicity characteristics leaching procedure
UC strength	unconfined compressive strength
ϕ'	internal friction angle, °

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Chapter 8

STABILIZATION OF BROWN COAL FLY ASH USING GEOPOLYMERS

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Abstract

Fly ash had been a problem for many power stations. This study has focused on using an encapsulation technique to stabilize the fly ash by a geopolymer. The potential of a geopolymer to stabilize fly ash was explored through the use of four geopolymer formulations. The leach rates of untreated fly ash helped to target of seven elements that are arsenic, barium, beryllium, selenium, strontium, calcium, and potassium. The lowest leach rates were observed at the geopolymer formulation with $\mathrm{Si/Al} = 2$.

Keywords: Fly ash, geopolymer, metal leaching, waste management, stabilization.

1. Introduction

Fly ash is a solid waste material produced from the burning of coal for electricity generation and has traditionally been seen as problem for power stations. Each year, millions of tons of fly ash are produced and disposed of in ash ponds and landfills. However, recently, fly ash has been seen as a resource[1,2] with many companies now looking at fly ash as a raw material with great potential for recycling and reuse. The composition of certain fly ashes, particularly those with high silica contents, makes them suitable for reuse in the construction industry as small percentages of addition to cement, bricks, road base, and other building products.[3,4] Research has also been conducted into novel ways of reusing fly ash such as mineral extraction[5–7] and zeolite synthesis.[8,9] The drawback in this scenario is the sheer volume of fly ash produced annually and the limited capacity of reuse that is available. This inevitably leads to many tons of fly ash still being stored in landfills.

Currently, fly ash in Australia is classified as a prescribed waste and its disposal in landfill is prohibited without stabilization or treatment. Trace heavy metals, such as mercury and lead, present in most fly ashes pose an environmental threat if the fly ash is landfilled without stabilization. The leaching of heavy metals into groundwater and surface water bodies could occur if the unstabilized fly ash comes into contact with water, resulting in serious environmental problems.

There are many methods available for the stabilization and treatment of fly ash to reduce the potential risks of heavy metal leaching. These include well-studied and documented examples such as acid washing,[10,11] cement fixation,[12–16] and thermal treatment by either sintering or melting.[17–21] Recently, a new technology has gained interest for waste stabilization called "geopolymer" stabilization and involves the encapsulation of the fly ash in a similar manner to cement fixation. The properties of these materials make them suitable for waste stabilization and include high strength, low permeability, high durability, high-temperature resistance, and excellent acid resistance. In the case of fly ash with high silica and alumina contents, geopolymer materials can be synthesized to achieve both to stabilize and to recycle the fly ash. The fly ash used in this chapter is of brown coal origin from the Latrobe Valley, Victoria, Australia. It has a unique composition in that it has a high calcium and magnesium content and low heavy metal content. Nonetheless, stabilization or treatment is still required, so that leach rates of all elements are reduced in order to achieve safe landfill disposal.

2. Background on Geopolymer Technology

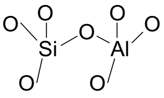
The term "geopolymer" was introduced by Davidovits in the early 1970s[22] and since then has gained general acceptance for materials of this nature, with many publications now referring to these aluminosilicate-based materials as geopolymers. Other common names for these particular materials include inorganic polymers and alkali-activated cements.

Geopolymers are an old technology that has recently received attention in various forms; one of which is waste stabilization. Geopolymers function in much the same way as a cement binder; however, as the reaction of cementation is one of hydration, the formation of a geopolymer network is one of polymerization. Instead of using calcium-silica-hydrate, as the building block of the structure, geopolymers require silica and alumina, commonly found in clays and minerals, in the presence of an alkali metal hydroxide. The manufacture and synthesis of geopolymers can be carried out from pure materials, such as clays (kaolin or metakaolin),[23,24] wastes rich in silica and alumina (mine tailing, builders waste,[25] fly ashes,[26–30] blast furnace slag[31,32]) or aluminosilicate minerals, and feldspars.[33,34] It is this versatile characteristic of geopolymers that makes them attractive to stabilize a variety of wastes.

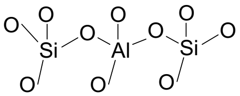
Basically geopolymer synthesis involves using materials with silica and alumina in the correct ratios and activated by an alkali metal hydroxide to produce a three-dimensional inorganic amorphous structure. The alkali metal hydroxide is generally sodium or potassium hydroxide, each giving the geopolymer different properties. Setting of the wet geopolymer mixture is carried out at temperatures up to 100°C, but curing at room temperature is possible and usually adequate.

As with all polymeric reactions, there exists a number of basic monomers that are present in solution from which larger unit (dimer, trimers, oligomers, and the network) are created during polymerization. The monomers participating geopolymer reactions include:

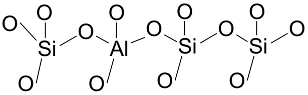
Sialate



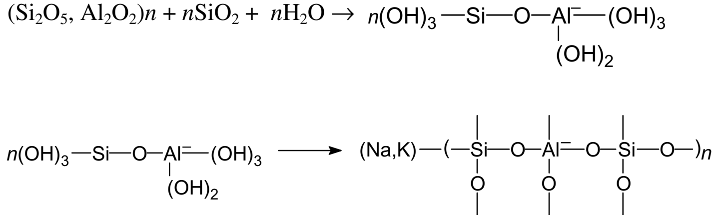
Sialate-siloxo



Sialate-disiloxo



The reaction mechanisms of geopolymers are very simple and involves the synthesis of aluminosilicate oxides (Al^{3+} in IV-fold coordination) with alkali polysilicates to give polymeric Si–O–Al bonds[22] according to the reaction in Fig. 1.



$$(Si_2O_5, Al_2O_2)n + nSiO_2 + nH_2O \rightarrow n(OH)_3{-}Si{-}O{-}\underset{(OH)_2}{\underset{|}{Al^-}}{-}(OH)_3$$

$$n(OH)_3{-}Si{-}O{-}\underset{(OH)_2}{\underset{|}{Al^-}}{-}(OH)_3 \longrightarrow (Na,K){-}({-}\underset{\underset{|}{O}}{\underset{|}{\overset{|}{Si}}}{-}O{-}\underset{\underset{|}{O}}{\underset{|}{\overset{|}{Al^-}}}{-}O{-}\underset{\underset{|}{O}}{\underset{|}{\overset{|}{Si}}}{-}O{-})_n$$

Figure 1. Polymerization of a Poly(Sialate-Siloxo) Geopolymer.[5]

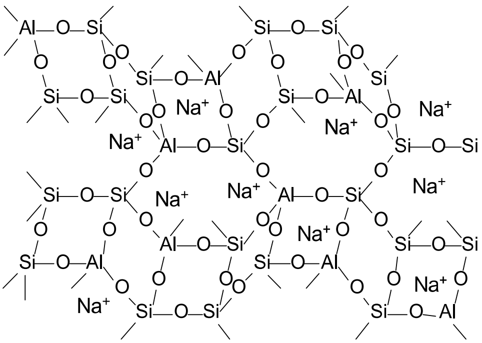


Figure 2. Sodium Poly(Sialate-Siloxo) Structure.[5]

The structure of the geopolymer network is based on the type of oligomers participating in the reaction. A poly(sialate-siloxo) geopolymer synthesized with sodium hydroxide and consisting of repeating Si–O–Al–O–Si units would have a polymeric chain and ring structure as shown in Fig. 2.

Stabilization properties of geopolymers can be improved by calcining clay prior to synthesis. Calcination removes the structural water of crystallization and results in a structural change that leaves Al^{3+} in IV-fold coordination.[35] This change makes aluminum more reactive and a polymeric material can be produced at room temperature, and without the need for elevated pressures. The presence of monovalent cations in the geopolymer network is required to balance the negative charge created by the aluminum in IV-fold coordination. This role is typically taken by the alkali metal hydroxide used to activate the geopolymer.

Precursors are produced from the dissolution of silica and alumina species from aluminosilicate sources. The dissolved silica and alumina oligomers polycondense to form a gel phase that constitutes part of the geopolymer. In order for polymerization to occur a filler material must be present and this role is taken by undissolved particles of the initial aluminosilicate raw material. Sand can also be specifically added as filler.[36] A geopolymer synthesized from metakaolin clay is

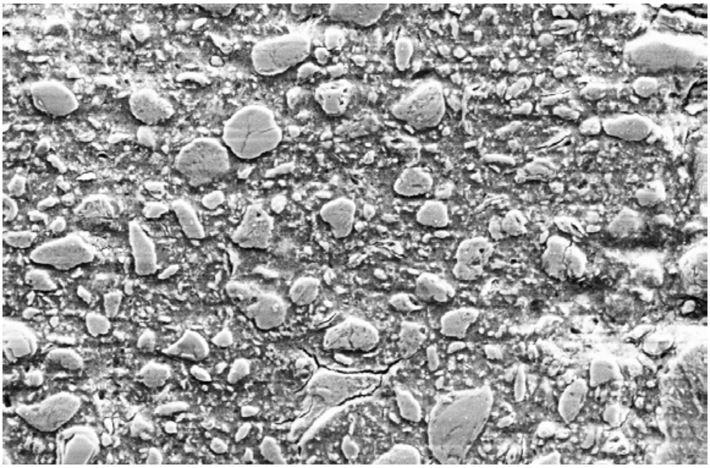


Figure 3. SEM Image of Pure Geopolymer.

shown in Fig. 3, demonstrating the filler material surrounded by the geopolymer gel phase.

Since waste particles can function both as part of the geopolymer structure and as a filler material there will be a limit on the capacity the matrix can tolerate before it becomes chemically or physically unstable and leaching levels become inappropriate. Davidovits[37] claims that pure geopolymers of potassium-poly(sialate-siloxo) type have the ability to encapsulate radioactive heavy metals with efficiencies of 90%+ based on an initial total concentration of 300 pCi/l. The maximum loading of a geopolymer structure was demonstrated by Van Jaarsveld *et al.*[38] with the synthesis of coal fly-ash-based geopolymers and observing leaching characteristics of lead and copper; however, a quantitative result on the maximum capacity of fly ash in the structure was not obtained. They noted that heavy metals stabilized by the structure have a tendency to leach more, as their concentration in the structure increases. Palomo[39] conducted leaching tests on fly ash spiked with lead and encapsulated in geopolymer, and found that the retention of lead in the matrix was very high. After 90 days of leaching, only 0.0255 mg of the initial 17.7781 mg had been released by the matrix into the solution and stabilization was attributed to the formation of Pb_3SiO_2.

Khalil and Merz[40] conducted experiments on the waste loading of geopolymers using synthetic radioactive wastes and found that a maximum loading ratio of 0.5 radioactive heavy metals to reactive component of the geopolymer was possible before the geopolymer failed to harden to its full potential. Since their reactive component was 30 wt% of the total geopolymer mass, addition of only 15% radioactive materials of the total geopolymer mass was possible.

3. Experimental Methods of Using Geopolymer to Stabilize Fly Ash

In this study, brown coal fly ash, collected from electrostatic precipitators of a Latrobe Valley power station, is used for experiments. The method used to stabilize the fly ash against leaching of heavy metals is outlined below with a description of sample preparation for leaching tests; scanning electron microscope (SEM) analysis and compressive strength tests.

The synthesis of geopolymers for the stabilization of fly ash can be approached via two pathways depending on the composition of the fly ash. Generally, fly ash that is high in silica and alumina can undergo geopolymer type reactions in the presence of an alkali metal hydroxide without the need for an added source of silica and alumina; however, the fly ash used in this study has low percentages of silica and alumina, as shown in Table 1. The major mineral phases as detected by x-ray diffraction (XRD) analysis (Fig. 4) were calcium sulfate, calcium oxide magnesium oxide, sodium sulfate, and sodium silicate.

Experiments have shown that simply adding a solution of alkali metal hydroxide (sodium hydroxide) directly to this fly ash did not form a geopolymeric material, so another approach was developed for successful stabilization to be achieved. This approach first focused on creating a geopolymer material and second adding the percentage of fly ash to be stabilized, since it was known that the fly ash alone would not form a geopolymer material. The focus of this research is then to stabilize the fly ash rather than the synthesis of geopolymerized fly ash materials. The fly ash stabilization procedure shown below was developed to ensure the formation of geopolymer and increase the potential for stabilization. This is just one possible method for the stabilization of fly ash using geopolymers. The process is a two-stage

Table 1. Composition of Raw Materials.

Compound	Precipitator Fly Ash (Mass %)	Kaolin Clay (Mass %)
SiO_2	2.4	49.3
Al_2O_3	1.2	35.0
CaO	30.6	0.1
MgO	16.4	0.3
Na_2O	4.3	0.2
Fe_2O_3	8.2	1.1
K_2O	0.3	0.2
TiO_2	—	1.1
SO_3	14.0	—
Loss on ignition (L.O.I.)	22.6	12.9

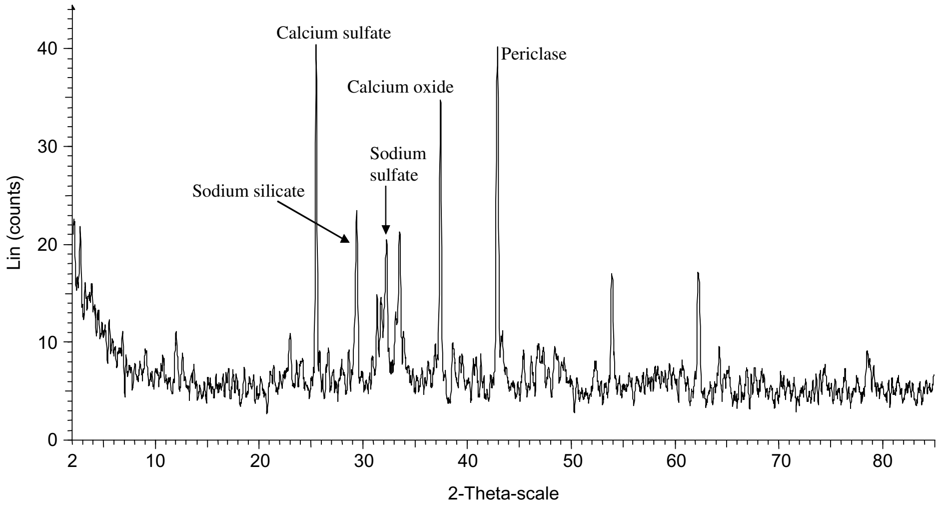


Figure 4. XRD Spectrum of Untreated Brown Coal Fly Ash.

process that minimizes chemical interaction between the fly ash and the geopolymer.

$$\begin{aligned}&\text{Clay} + \text{sodium silicate solution} + \text{sodium hydroxide} + \text{water}\\&\quad\rightarrow \text{geopolymer paste geopolymer paste} + \text{fly ash} \rightarrow \text{stabilized fly ash}\end{aligned}$$

The formulation of the geopolymer used throughout the stabilization experiments has a molar silica to alumina ratio of two and three. Geopolymers that are suitable for waste stabilization have a molar silica to alumina ratio of two.[41] The clay used was kaolin clay with the composition as shown in Table 1, and had a molar silica to alumina ratio of one, so silica was added in the form of sodium silicate solution to increase the silica to alumina ratio. The effect of the silica and alumina content in the fly ash on the geopolymer formulation was ignored since the percentage of silica and alumina in the fly ash is small compared to the mass in the geopolymer and also not all silica and alumina in the fly ash will dissolve during synthesis.[42]

A total of four geopolymer formulations were used to stabilize fly ash. The first formulation (Formulation 1) involved encapsulation in a geopolymer with a silica to alumina ratio of two. This formulation produced a geopolymer with some inherent processing difficulties, such as pressing of the uncured materials and reduction in pH when high percentages of fly ash were added, and so the formulation was altered to minimize these problems. Formulation 2 involved the addition of water into the fly ash prior to mixing with the geopolymer paste that eliminated the need for pressing of the uncured material. Formulation 3 was developed to compensate for the reduction in pH of the geopolymer paste as the fly ash content was increased and focused on the addition of increasing percentages of sodium hydroxide as the percentages of fly ash were increased, maintaining a constant pH in the geopolymer paste. Finally, Formulation 4 was calculated that attempted to stabilize the fly ash in a geopolymer with a silica to alumina ratio of three.

The water content of Formulations 1, 2, and 3 was held constant at 46 wt% of the geopolymer phase that is why the mixture transformed from a paste into a powder with increasing addition of fly ash and required pressing to consolidate. Formulation 4 had a water content of 43 wt% of the geopolymer phase. Extra water in Formulation 2 was added to the fly ash prior to blending with the geopolymer, to preserve the chemistry of the geopolymer during activation and synthesis.

The formulations were calculated to have the molar ratios as shown in Table 2. The ratio of sodium to aluminum must be greater than or equal to one, so that the charge of each aluminum atom in IV-fold coordination is balanced by a sodium ion.[22] In geopolymers synthesized from less pure materials, molar ratios are not as critical since other ions; for example, potassium and calcium can fulfill this role. The molar ratios of Formulation 3 are not shown since these varied with each sample and were dependent on the pH of the paste and the percentage of fly ash added to the sample.

In order to gain a better understanding of the potential of stabilization of the geopolymer toward heavy metals, a series of spiked samples was prepared. The chosen heavy metals were mercury, lead, barium, and zinc. Three samples in total were spiked with the heavy metals and two contained fly ash. The samples prepared were pure geopolymer spiked with 2 wt% of each heavy metal ion, geopolymer, and fly ash spiked with 2 wt% heavy metal ion and geopolymer and fly ash spiked with 0.6 wt% heavy metals as shown in Table 3. Samples were formulated such that the total heavy metal and fly ash content did not exceed 30 wt%.

Table 2. Geopolymer Formulation Molar Ratios.

Ratio	Formulation 1	Formulation 2	Formulation 4
Na/Si	1.8	1.8	1.4
Si/Al	2.0	2.0	3.0
Na/Al	1.1	1.1	2.0

Table 3. Spiked Sample Formulations.

Sample	Fly Ash Content (wt%)	Heavy Metal Content (Grams of Each Heavy Metal Ion)
0.6 wt% each (with fly ash)	17.1	3
2 wt% each (with fly ash)	5.3	15
2 wt% each (without fly ash)	0.0	15

The procedure for the synthesis of a geopolymer for stabilization of fly ash is described below.

Step 1. Metakaolin clay was produced by calcining kaolin clay at 750°C for 10 h.

Step 2. An activator solution consisting of sodium silicate solution, sodium hydroxide, and water was prepared by weighing all components and mixing. Sodium hydroxide was obtained in pellet form and distilled water was used. The sodium silicate solution had a molar silica to sodium oxide ratio of two. The activator solution was allowed to mix until all sodium hydroxide had dissolved.

Step 3. Metakaolin was slowly added to the activator solution whilst mixing to avoid the formation of lumps. Once all the metakaolin had been added the paste was allowed to mix for a further 10 min to promote silica and alumina dissolutions and monomer formations.

Step 4. After mixing, fly ash or heavy metals were added to the paste and blended until uniform.

Step 5. Depending on the fly ash content, the paste was either poured into molds and cured, or pressed to 10 MPa in specially designed press and then cured. All curing was carried out at 65°C for 24 h. The material was then removed from the mold and allowed to cure for a further 6 days at room temperature.

Varying percentages of fly ash were added to the geopolymer, in 10 wt% increments, to determine the point where leaching becomes excessive and the geopolymer cannot encapsulate the fly ash. At each percentage, leaching tests and SEM imaging were used to determine the leaching and interaction of the fly ash with the geopolymer. XRD and magic angle spinning nuclear magnetic resonance (MAS-NMR) were also used to study the stabilized fly ash.

4. Analytical Tools used to Study the Effectiveness of Stabilization

The effectiveness of stabilization was determined using the standard USEPA TCLP (toxicity characteristic leaching procedure) leaching test.[43] Sample preparation for the leaching test involved crushing of the cured samples to yield pieces smaller than 9.5 mm. The crushed samples were then leached for 18 h in dilute acetic acid solution. The leachate was centrifuged to separate particles from the solution and the supernatant was filtered using vacuum filtration with a 0.45-μm filter. From the filtered leachate, 250 mL were transferred into glassware and acidified with 4 mL of concentrated (16 M) nitric acid. This produced a pH of less than two for all leachate samples. The samples were then stored under refrigeration until analysis.

The main tool for the analysis of the leachate solutions was inductively coupled plasma optical emission spectrometry (ICP-OES). Initially, a small portion of samples were analyzed using ICP mass spectrometry (ICP-MS) for a wide range of elements to determine which required further attention and which could be eliminated because their concentrations were below detection limits. The final list was reduced to arsenic, barium, beryllium, strontium, and selenium. It was found that toxic elements such as lead, mercury, chromium, and cadmium were present below the detection limits in the leachate solutions and could be eliminated from further analysis. The remaining analysis was conducted using ICP-OES.

Samples were analyzed in batches of 10–15 samples with standards and calibration being performed between each batch. This reduced errors associated with machine drift over long periods and allowed for calculations to be performed that accounted for these errors. Usually, the most sensitive wavelength was chosen for each element in the analysis program unless interference was observed from other elements. In this case, the next sensitive wavelength was chosen.

The interaction of the fly ash with the geopolymer was studied using SEM (Leo 1530) and involved the procedure mentioned earlier for the synthesis of the fly ash-geopolymer samples and well as a mounting and preparation process. Samples were mounted in low viscosity Spurr's resin, used in biological sample SEM mounting because of superior sample penetration. Standard epoxy resin used in metallographic mounting causes samples to "tear away" during grinding and polishing resulting in scratches and poor images. Each sample was shaped manually, placed in a mold filled with Spurr's resin and subjected to vacuum for 8 h to remove air bubbles. Oven curing overnight completed the curing of the mold and produced a stable mounted sample.

Mounted samples were initially ground with 500 grit silicon carbide paper to expose a cross section of sample then slowly progressed through 800 and 1200 grit. The next stage was diamond suspension polishing beginning with 15 μm suspension and finishing with 1 μm suspension. Prior to polishing and between each stage of polishing ultrasonic cleaning in water was employed to remove large particles and prevent contamination of samples and polishing pads. The final stage of cleaning was done in ethanol and the samples were surface dried then placed in a desiccator until required for analysis.

Samples for SEM imaging were carbon coated and painted with silver paint to improve conductivity and minimize charging of the sample surface. Typical microscope operating conditions are shown in Table 4 and were chosen for clarity of images and well as accuracy of analysis. Energy dispersive x-ray (EDX) analysis was also carried out on selected samples; therefore, the chosen conditions allowed fast transition between imaging and EDX analysis without the need for reconfiguration of the microscope.

Table 4. Typical SEM Operating Conditions.

Parameter	Value
Chamber vacuum	$1 \times 10 - 5$ mbar
EHT	20 KV
Aperture	30 μm
Working distance	15 mm
Spectrum acquisition time	150 s
Elemental map dwell time per pixel	150 ms

EDX elemental maps were used to determine the distribution of five elements across the surface of the sample. The chosen elements were calcium, magnesium, sodium, silicon, and aluminum. The generation of an elemental map was also accompanied by the collection of compositional data of the entire mapped region as well as certain features of the map; for example, particles, spheres, light and dark regions. This was used to determine whether an area was suitable for mapping.

Potential applications of the stabilized fly ash-geopolymer materials was determined by conducting compressive strength tests in general accordance with AS 2350,[44] concrete compressive strength testing procedure. Samples were prepared as described previously and poured into 40 × 40 × 160 mm molds for curing. Those that required pressing were pressed into 40 × 40 × 40 mm cubes at 10 MPa and cured. The average compressive strength was calculated from the compressive strength of several samples. The number of samples was reduced to make the experiments more practical and in general two to four samples were tested rather that four to six.

5. Results of Stabilization of Fly Ash by Geopolymers

5.1. *Evidence of Geopolymer Formation*

Fly ash collected from electrostatic precipitators was stabilized in a geopolymer material. First, it was established whether a geopolymer was indeed forming using the described synthesis procedure or perhaps the material was consolidated by other means.

The formation of pure geopolymer was confirmed by NMR analysis (Fig. 5) and typical 29Si MAS-NMR features were observed that agree with other published information on the structure of geopolymers.[22,45,46] Broad resonance, associated with a disordered structure, was observed at −99.6 ppm. Other peaks were detected at −102 ppm and −97.5 ppm. The peaks correspond to $SiQ_4(2Al)$, $SiQ_4(3Al)$ and $SiQ_4(1Al)$ bonding, respectively.

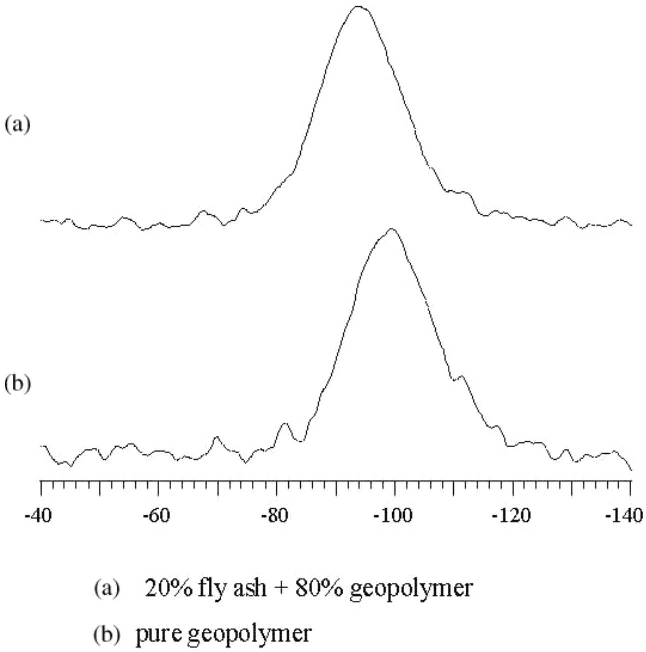


Figure 5. 29Si MAS-NMR Spectra of Pure Geopolymer and 20 wt% Fly Ash–80 wt% Geopolymer.

It was found that the introduction of fly ash into the geopolymer caused a slight shift up-field in the spectrum from that observed in the pure geopolymer. The broad resonance peak for 29Si MAS-NMR is observed at -94.4 ppm instead of -99.6 ppm, although a change in shape of the spectrum was not observed. Peaks that corresponded to $SiQ_4(3Al)$, $SiQ_4(2Al)$, and $SiQ_4(1Al)$ have become less resolved in the fly ash-geopolymer sample indicating that perhaps the structure is more disordered, due to the presence of fly ash, than that observed in pure geopolymer.

5.2. *Stabilization of Trace Metals and Major Elements in Fly Ash*

Leaching tests were used as the primary evaluation technique for the stabilization of brown coal fly ash using geopolymers. Trace metals were given priority in terms of stabilization, due to their toxicity, and emphasis was placed on reducing their leach rates. Literature has focused on the stabilization of some heavy metals such as lead and copper, stabilized in fly ash-based geopolymers[47] rather than pure geopolymers; however, Davidiovits[37] has carried out research into the stabilization of heavy metals in pure geopolymers for commercial purposes with very effective results.

In Table 5, trace metal analysis from fly ash-geopolymer samples is given for the trace metals that were initially established as being present in high concentrations and potentially problematic. These metals also showed a clear decrease in leach

Table 5. Trace Metal Analysis from Fly Ash-Geopolymer Material (μg/L).

Percentage Fly Ash in Geopolymer	As	Ba	Be	Se	Sr
Untreated	12	270	16	740	31,200
60	2	120	n.d.	110	11,500
50	n.d.	110	2	100	9,900
40	2	70	n.d.	130	5,500
30	2	80	n.d.	110	4,500
20	1	80	n.d.	60	2,400
10	1	70	n.d.	60	1,100

Note: n.d. = Not detected.

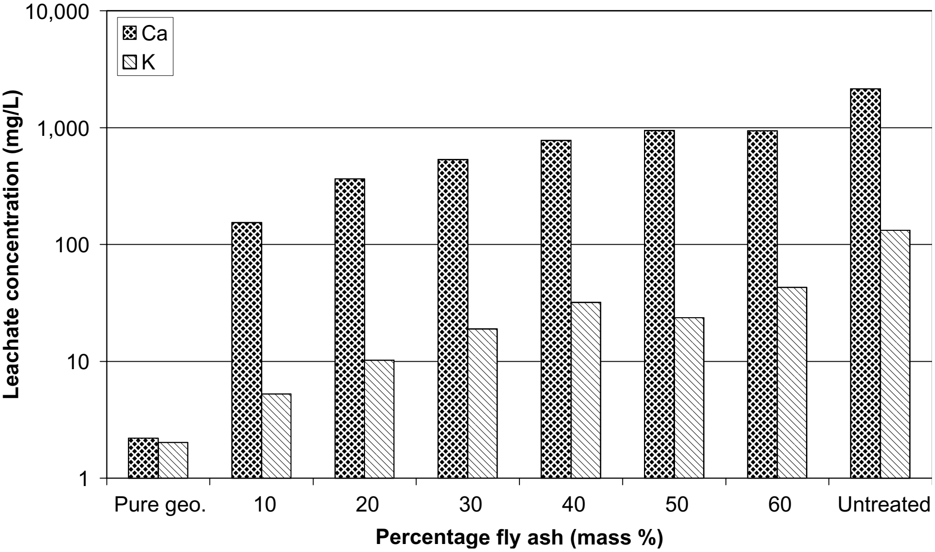


Figure 6. Major Element Leachate Analysis from Fly Ash-Geopolymer Material with Varying Amounts of Fly Ash.

rates when fly ash was stabilized by the geopolymer. The leach rate of strontium has decreased considerably from 31200 μg/L in untreated fly ash to 1100 μg/L when stabilized by 90 wt% geopolymer. The leach rate of selenium also decreased substantially from 740 μg/L in untreated fly ash to 60 μg/L when fly ash was added at 10 wt% to the fly ash. Arsenic, barium and beryllium, although initially present in lower percentages in the untreated fly ash, still showed reductions in leach rates.

The stabilization of major elements was given less priority since these are less toxic in nature and generally comprise of calcium. From the fly ash leachate data in Fig. 6, there is a general trend that shows a geopolymeric structure has the ability to reduce the leaching rate of major elements, particularly at low percentages of fly ash. Calcium and potassium leach rates were reduced considerably for fly ash since these elements can participate in the geopolymer network in charge balancing roles. The leach rate of calcium from untreated fly ash was determined to be approximately 2100 mg/L and was reduced to 150 mg/L with the addition of 90 wt% geopolymer. Potassium showed similar reduction from 130 mg/L to 5 mg/L at the same percentages.

The maximum addition of a waste into the geopolymer matrix must be established in order to reduce the mass of geopolymer reactants required, and hence cost, and to determine the leach rates at maximum addition. Since this fly ash largely does not participate in geopolymer network formation, the maximum addition can be found by progressively increasing the fly ash addition until leaching becomes inappropriate or the stable matrix fails to form.

Using the geopolymer formulation with Si/Al = 2, the maximum addition of fly ash to the geopolymer was 60 wt%, based on observations during leaching tests. Experiments have shown that the geopolymer phase is insoluble and any geopolymer formed would not dissolve when placed in water. Therefore, after the leaching test any remaining material is stable geopolymer. If no consolidated material remained in the leaching vessel, then it was assumed that not enough geopolymerization had occurred to bind the fly ash. Addition of 60 wt% fly ash produced a minimal amount of geopolymer material after the leaching test. Increasing the fly ash content in the geopolymer by another 10 wt% did not produce any remaining geopolymer after leaching. The structural integrity of these samples was reduced such that they dissolved during the leaching test, indicating that minimal geopolymer formation had originally taken place and solidification was not achieved. In fact, the reduction in structural integrity was observed at 50 wt% fly ash, but small fragments of geopolymer stabilized fly ash still remained after the leaching test.

5.3. *Additional Water in the Geopolymer Formula*

The addition of water to the fly ash, as described in formulation 2, caused an increase in the overall leach rates of the geopolymer stabilized fly ash (Fig. 7). Less fly ash is added to each sample to accommodate the water, so that the geopolymer content for each sample remained the same as that in previous experiments. As a consequence of the decreased fly ash addition and increased leach rates, the leach rates per unit mass of fly ash must increase. In Fig. 7, the leach rates of calcium and potassium are shown. The leach rates of calcium and potassium were reduced to 122 mg/L and 3 mg/L, respectively, however, at lower percentages of fly ash. These leach rates were observed at 5.7 wt% fly ash addition to the geopolymer. At 10 wt% fly ash, the leach rates would be approximately 270 mg/L for calcium and 5 mg/L for potassium,

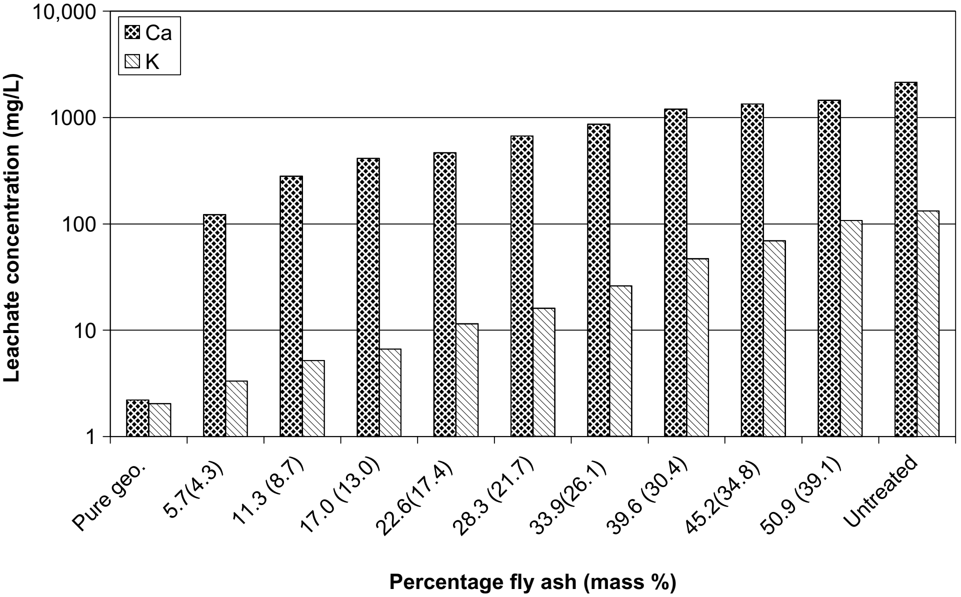


Figure 7. Calcium and Potassium Leach Rates From Fly Ash-Geopolymers with Fly Ash/Water = 1.3. Values in Brackets Indicate Percentage of Water in Fly Ash.

Table 6. Trace Metal Leach Rates in Fly Ash-Geopolymer with Fly Ash/Water Ratio = 1.3 (μg/L).

Percentage Fly Ash in Geopolymer	As	Ba	Be	Se	Sr
Untreated	12	270	16	740	31,200
45.2	n.d.	114	4	87	11,026
33.9	n.d.	77	3	94	6,710
22.6	n.d.	69	n.d.	80	3,108
11.3	n.d.	64	n.d.	45	1,566

Note: n.d. = Not detected.

which is higher in terms of calcium compared to stabilization by the geopolymer with Si/Al = 2.

Trace metals were analyzed for every second sample (Table 6) since major elemental leachate analysis showed that stabilization was slightly less effective with the addition of water. In general, there is a trend for reduction of all elements in Table 6, which is very encouraging. The leach rates of selenium and strontium have been reduced to 45 μg/L and 1566 μg/L at 10 wt% fly ash addition, respectively. Arsenic, barium, and beryllium also showed reduction in leach rates that are comparable to those of the geopolymer with Si/Al = 2. Comparison of the fly ash-geopolymers with increased water content (Formulation 2) to stabilization by geopolymers with Si/Al = 2 (Formulation 1) shows similar results as observed in the latter. This suggests that the same mechanisms are operating to reduce leaching regardless of the water content.

Maximum addition of fly ash to this formulation was achieved at 50.9 wt%. The difference between the maximum addition for this formulation and the geopolymer with Si/Al = 2 is the percentage of water added to the fly ash. The water replaced a portion of the possible fly ash addition and reduced the maximum to 50.9 wt%. This is a mathematical limitation as opposed to a physical maximum achieved experimentally and possibly a reason to reduce the water content of the formulation.

The increase in leaching of some elements is a direct result of the water added to the fly ash prior to mix with the geopolymer. The water cannot be added to the geopolymer since a lowering of the pH in the geopolymeric solution would result in poor precursor formation and lead to poor geopolymerization. The water added to the fly ash partially dissolves the particles of the ash and creates a solution of metal cations. Theoretically, certain cations are able to participate in geopolymeric reactions by lending themselves to change balancing roles. This can occur to a certain extent; however, as the percentage of fly ash and water increases, the geopolymer becomes saturated with cations and stabilization is reduced.

Retaining the particles of the fly ash as a whole particle would appear to be the better mechanism of stabilization. Compounds in fly ash particles completely surrounded by geopolymer must diffuse through a physical barrier, making the whole process more difficult. In addition, any material on the inside of particles must also diffuse through particle itself before reaching the geopolymer. Bonefede[48] has stated that particles of brown coal fly ash are not of uniform composition; therefore, the presence of a physical barrier to diffusion is an important characteristic of stabilization.

5.4. *Additional Sodium Hydroxide in the Geopolymer Formula*

Geopolymers rely on the presence of an alkali metal hydroxide to dissolve clay particles during synthesis and also balance the charge created by aluminum in the geopolymer network. According to literature, increasing the sodium hydroxide content will promote more dissolution of geopolymeric species and in turn increase the strength of the geopolymer, since more dissolution leads to more monomers/oligomers and better network formation. This being the case, introducing more sodium hydroxide in this fly ash-geopolymer system would lead to better stabilization through chemical and physical means. More dissolution of aluminum would create more sites where cations from the fly ash could bond (substituting sodium) and also create a better physical barrier through greater bonding of oligomers. This forms the basis of Formulation 3.

Table 7. Trace Metal Leach Rates from Fly Ash-Geopolymer with Extra Sodium Hydroxide (μg/L).

Percentage Fly Ash in Geopolymer	As	Ba	Be	Se	Sr
Untreated	12	270	16	740	31,200
60	n.d.	74	3	114	7,946
50	n.d.	64	n.d.	n.d.	6,548
40	n.d.	69	n.d.	n.d.	5,231
30	n.d.	66	n.d.	n.d.	3,661
20	n.d.	77	2	51	2,443
10	n.d.	97	n.d.	88	1,799

Note: n.d. = Not detected.

The results of the above assumption can be seen in Table 7. As with previous data there is a trend toward the reduction of the concentration of ions in the leachate. The two elements with the highest leach rates from untreated fly ash, selenium and strontium, showed good results with a reduction in leach rates from 740 μg/L to 88 μg/L for selenium and from 31200 μg/L to 1799 μg/L for strontium, both at 10 wt% fly ash. For this formulation, the stabilization of barium was less effective particularly at low percentages of fly ash. The leach rates of arsenic and beryllium also remained low for this formulation. By comparing the leach rates with the best available results, those of geopolymers with Si/Al = 2 stabilized fly ash, it can be seen that similar trends exist in terms of the behavior of trace and major elements.

Trace metals in Table 7 responded to stabilization with reduction in leach rates of all five elements over that of untreated fly ash. When compared to the leach rates of the geopolymer with Si/Al = 2 (Table 5), these leach rates are marginally lower for barium and convincingly lower for strontium and selenium. They are, however, higher for beryllium. The leach rates at low percentages of fly ash, less than 20 wt%, increase for selenium and barium and are higher than those of fly ash stabilized by geopolymers with Si/Al = 2. This formulation has similar leaching results to the geopolymer with Si/Al = 2 with a greater leach rate reduction for fly ash contents greater than 30 wt% but lower reductions and even increases for lower percentages of fly ash.

The leach rates of major elements for fly ash-geopolymers (Fig. 8) showed similar trends to previous formulations, in particular that of geopolymers with Si/Al = 2. The leach rates of calcium and potassium are reduced throughout the range of fly ash addition. However, the magnitude of the leach rates is higher than geopolymers with Si/Al = 2, particularly at percentages greater than 40 wt% fly ash. The participation of calcium, potassium, and sodium in the geopolymer network has already been established. If the concentration of sodium in the geopolymer is increased, then fewer

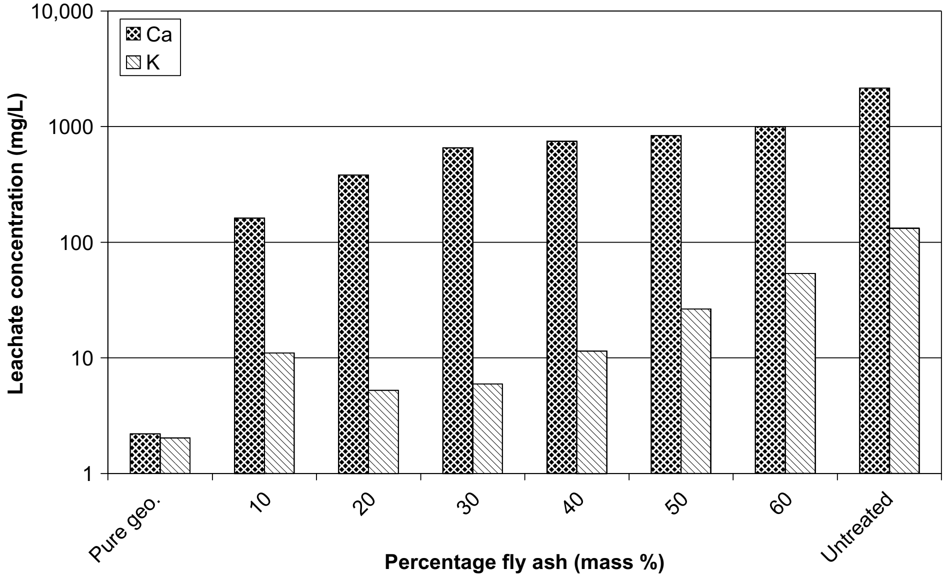


Figure 8. Major Element Leachate Concentrations from Fly Ash-Geopolymers with Increased NaOH Content.

sites are available for calcium and potassium to bind resulting in higher leaching of these elements.

The initial assumption was that the addition of sodium hydroxide will produce a better network through dissolution of more silica and alumina and create more sites for chemical stabilization. On a large scale, this has not occurred and the stabilization of heavy metals was improved at fly ash percentages of 30 wt% or greater, with the poorer stabilization at lower percentages. The leaching of calcium has increased compared to that achieved by geopolymers with Si/Al = 2. Therefore, it appears that the addition of sodium hydroxide has a mixed effect on stabilization of fly ash.

The use of less sodium hydroxide in the geopolymeric solution could be possible since the clay can be activated to a certain degree in solution even with low concentrations of sodium hydroxide. Charge-balancing roles can be fulfilled by cations in the fly ash once blending has been carried out. The strength of the fly ash-geopolymers will be compromised since the more the clay is activated, the high the strength of the final material will be. Although strength is not considered as an important property of the stabilized fly ash, some strength is required to withstand handling, disposal, to consolidate the material and provide a physical barrier to leaching. In general, the concentration of sodium hydroxide could possibly be reduced to save raw materials.

The addition of sodium hydroxide to maintain a constant pH in the geopolymer-fly ash paste had no effect on the maximum addition of fly ash to the geopolymer.

The maximum possible addition was limited at 60 wt% and is equal to geopolymers with Si/Al = 2 and greater than geopolymer with extra water. For this formulation, samples containing 50–60 wt% showed poor structural integrity and had little geopolymer material remaining after the leaching test indicating that fly ash addition was nearing a maximum.

5.5. *Geopolymer Formula with Si/Al ratio of Three*

Although good leaching results were obtained for geopolymer stabilization when the silica to alumina ratio was two, stabilization with geopolymers having a molar silica to alumina ratio of three was also explored (Formulation 4). Theoretically, this Si/Al = 3 geopolymer is better suited to high-temperature applications such as foundry equipment and fire protection composites; however, with the possibility of consumption of some silicate by side reactions with fly ash, the ratio in the geopolymer phase is reduced to below three. Indeed, in Si/Al = 2 geopolymers, the ratio is reduced below two.

In general, there is a trend toward a reduction in leaching because of the encapsulation by the geopolymer material with Si/Al = 3, although not as clear as those of previous formulations, with some elements showing increases, as the percentage of fly ash is reduced. In Table 8, trace metal analysis from fly ash-geopolymer samples are given. Those elements that have shown good stabilization for the previous three formulations have shown good results for this formulation, in particular strontium. The leach rates of selenium, barium, and beryllium were reduced; however, the leach rate of arsenic increased. It appears that the increased silica content of this formulation adversely affects the leaching of arsenic and its leach rates increased over those of unstabilized fly ash.

Table 8. Trace Metal Leaching from Fly Ash-Geopolymer Materials with Silica/Alumina Molar Ratio of three (μg/L).

Percentage Fly Ash in Geopolymer	As	Ba	Be	Se	Sr
Untreated	12	270	16	740	31,200
80	35	84	n.d.	94	14,404
70	2	68	n.d.	112	11,053
60	19	56	n.d.	156	8,068
50	62	12	n.d.	120	4,830
40	225	14	n.d.	84	4,632
30	67	18	n.d.	82	3,472
20	32	41	n.d.	148	2,242
10	240	49	n.d.	193	888

Note: n.d. = Not detected.

The stabilization of trace metals in fly ash-geopolymers was less effective for this formulation than for geopolymers with Si/Al = 2. Fly ash-geopolymers with Si/Al = 2 showed good stabilization of arsenic, barium, beryllium, selenium, and strontium, whereas the stabilization by geopolymers with silica to alumina ratio of three showed good results for the same elements except arsenic (Table 8).

Pure geopolymer of silica to alumina ratio of three used in this study failed to cure. Excess water present from the sodium silicate solution diluted the reactants in the geopolymer paste and the lack of catalyst normally provided by sodium increased the curing times dramatically. Certain ratios have to be met in terms of sodium/silica ratios that were not in this case. In short, this formulation has been designed with fly ash addition in mind to promote the curing process. Consequently, pure geopolymer is not represented in leaching data.

Major elemental leach rates are reduced, as observed in Fig. 9. Significant reductions have occurred for calcium and potassium in samples of both ash types. The leach rates of this formulation, although being low, are not quite as low as those observed in geopolymer stabilization with Si/Al = 2. In general, there is little difference between the Formulations 1 and 4 in terms of leaching.

The water present in the sodium silicate solution increased the maximum addition of fly ash to 80 wt% for both ash types. Compared to the other formulations, this is 20 wt% more fly ash than Formulations 1 and 3. It is also 29.1 wt% more than that

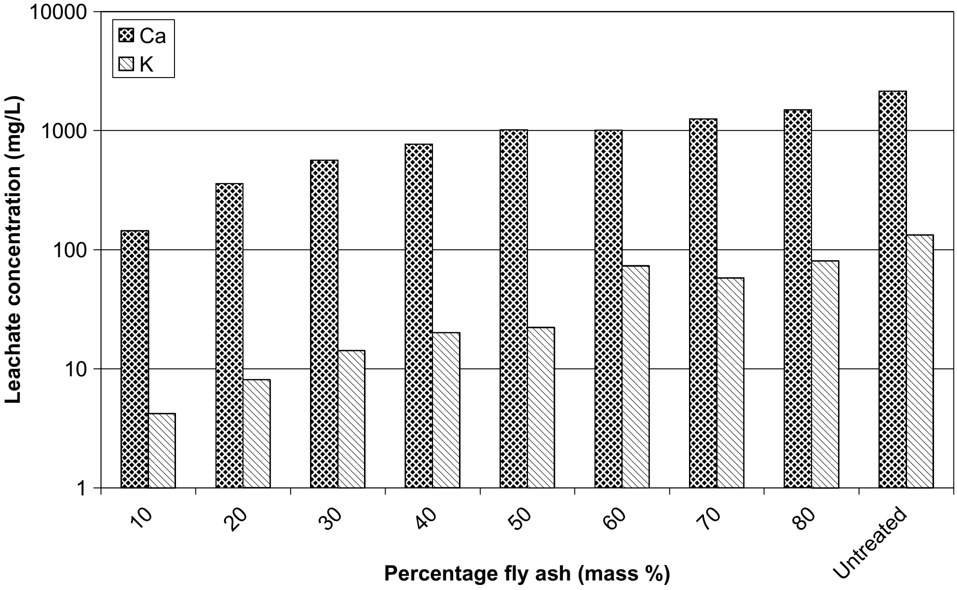


Figure 9. Major Element Leachate Concentrations from Fly Ash-Geopolymers with Silica/Alumina Molar Ratio of Three.

of Formulation 2. It must be noted that the structural integrity of the samples was very poor for samples containing 70–80 wt% fly ash.

The main reason for the increase in silica content of the geopolymer was to promote the formation of silicates and stabilize the fly ash by producing precipitates as well as stabilization by the geopolymer network. The leaching data suggests that this was not the case since a marginal increase in leaching was observed, compared to geopolymers with $Si/Al = 2$ and reinforcing that geopolymers with a molar silica to alumina ratio of two are best suited to waste stabilization.

5.6. *Fly Ash-Geopolymer Matrix Spiked with Heavy Metals*

To better understand the leaching characteristics of the trace elements, three different samples were artificially spiked with heavy metals. A total of four heavy metals were added to the spike. Barium was chosen because it showed good stabilization by the geopolymer and mercury, zinc, and lead were chosen because their concentrations were below detection limits and the response to stabilization could not be established from leaching data alone. It was anticipated that SEM/EDX analysis would give a better indication of the stabilization of mercury, zinc, and lead even though initial leaching data is inconclusive. Whether mercury, zinc, and lead are stabilized, resulting in low leaching, or unstabilized, resulting in free leaching, could not be determined from leaching data. The use of SEM/EDX could answer this question from the distribution of these elements and, later, leaching tests.

The first sample contained a combination of fly ash and heavy metals spiked to 0.6 wt% of the total sample mass. This gave 2 wt% heavy metals based on fly ash content that was 30 wt% of the total sample mass. The second sample contained heavy metals each of 2 wt% calculated based on total mass of sample. Fly ash was also added to this sample such that the total waste content was 30 wt%, resulting in 70 wt% geopolymer. The third sample simply contained geopolymer spiked with four heavy metals each of 2 wt% based on the total mass. The mix design of the samples was carried out in such a way that the samples could still be cured as a paste, with the geopolymer as the dominant phase.

The three samples were subjected to the leaching test and the results are tabulated in Table 9. The heavy metal content was calculated, based on the percentage of the metal ion rather than the compound. The actual mass of each compound is, therefore, greater. Fly ash was added to two samples to determine what effect it would have on the leaching of heavy metals. The effect of fly ash on heavy metal stabilization could be determined and then related back to the actual heavy metals in the fly ash itself.

Table 9 shows that given the mass of heavy metals in each sample, the geopolymer has effectively prevented the leaching of the spiked heavy metals. When the spiked

Table 9. Heavy Metal Stabilization by Geopolymer.

Heavy metal content		Pb	Hg	Ba	Zn
0.6 wt% each (with fly ash)	Leach rate (mg/L)	0.134	3.098	0.048	1.811
	% Retained	99.994	99.862	99.998	99.920
2 wt% each (with fly ash)	Leach rate (mg/L)	17.575	471.850	0.001	818.611
	% Retained	99.766	93.709	99.999	89.085
2 wt% each (without fly ash)	Leach rate (mg/L)	6.031	393.682	0.156	29.885
	% Retained	99.920	94.751	99.998	99.602

heavy metals were added to the fly ash-geopolymer at 0.6 wt%, each the leach rates remained less that 2 mg/L for lead, barium, and zinc. The leach rate of mercury was highest, although still low, at 3.098 mg/L. Increasing the heavy metal content to 2 wt% each increased the leach rates of all elements except barium that showed a slight decrease. The addition of fly ash to the system caused an increase in the leach rates of lead, mercury, and zinc and a slight decrease in barium. It is not until the mass of each heavy metal is compared to the mass percentage retained in the sample that the stabilization effect of the geopolymer is proven. Table 9 also shows the mass percentage retained by the geopolymer, assuming that all spiked heavy metals are leachable. In most cases, the percentage retained is over 99%. Given the increase in mass of heavy metals in the first two samples in Table 9 was from 3 g to 15 g, the decrease in the mass percentage of the heavy metals retained is small and high percentages are still retained by the fly ash-geopolymer.

6. Mechanisms of Interaction between Fly Ash and Geopolymer

The mechanisms of binding heavy metals in the structure of the geopolymer are not fully understood. There exists some speculation as to whether the binding is chemical with species taking part in the matrix or having charge balancing roles or physically being encapsulated in the matrix. Van Jaarsveld *et al.*[44] suggested that immobilization is largely based on a physical principle, since materials with smaller original particle sizes and hence smaller pores showed a decrease in leach rates. However, this was offset by the diffusion of metal species from the surface of particles leading to the conclusion that the mechanism is surface controlled. They also noted that metals not taking part in the structure weakened and destabilized both the inter- and intratetrahedral structure of the geopolymer making the material prone to leaching.

The addition of fly ash produces a more disordered structure in the geopolymer, as shown by MAS-NMR, which makes stabilization less effective. Current stabilization theory for geopolymeric materials centers on chemical and physical means. Chemical stabilization has been hypothesized as cations in charge-balancing roles and covalent bonds to the geopolymer structure via silica. Using this theory as a basis, the partial dissolution of fly ash produces cations that must be stabilized chemically and reduces the effective number of sites available for chemical stabilization. These sites are for charge-balancing and covalent bonding allowing cations to attach themselves to the geopolymer structure. The stabilization of the heavy metals in pure geopolymer showed lower leach rates since there was less competition for these chemical stabilization sites. That is not to say that physical stabilization is not occurring. Compounds that must be physically stabilized still form, however, the leach rates of these are unlikely to increase dramatically, since the geopolymer percentage is constant in all samples.

Before the stabilization theory can be applied to these samples, it must be made clear that each sample size was limited and the results are only meant to be used a stepping stone to the determination of the stabilization mechanism of each element analyzed for this fly ash-geopolymer system. The research was conducted to simply better understand the behavior of certain elements in this fly ash-geopolymer system and not to develop mechanisms of stabilization.

SEM was used to map the distribution of the heavy metals in the spiked samples. The sample with 2 wt% heavy metals and fly ash was chosen. The secondary electron image of the mapped region is shown in Fig. 10 and the elemental maps can be seen in Fig. 11.

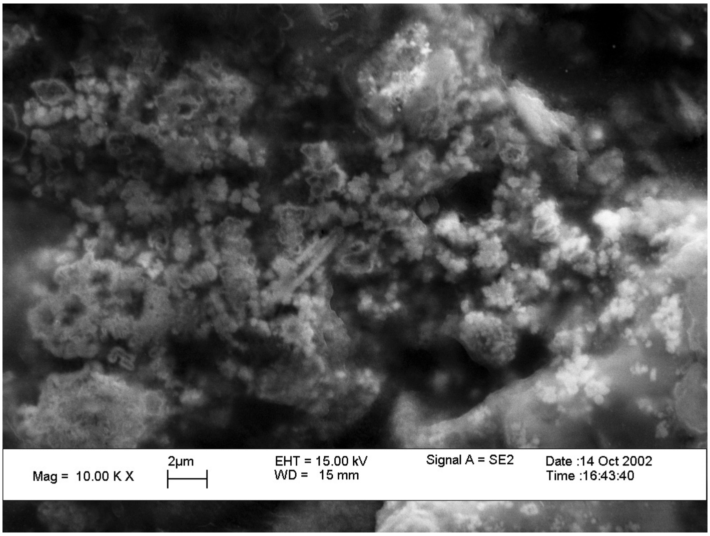


Figure 10. SEM Image of Heavy Metal Spiked-Fly Ash-Geopolymer.

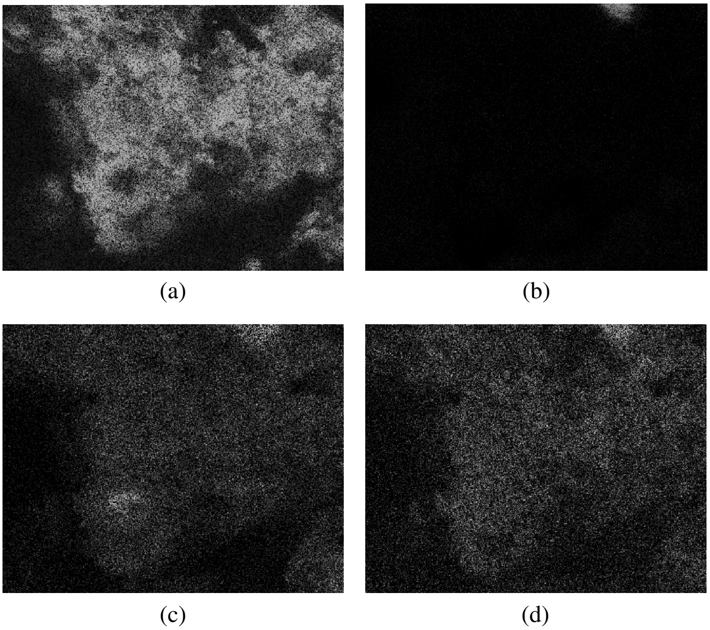


Figure 11. Distribution of Heavy Metals in 30 wt% Fly Ash, 2 wt% of Total Mass Heavy Metal Spiked Sample; (a) Barium distribution, (b) Mercury distribution, (c) Lead distribution, and (d) Zinc distribution.

The distribution of zinc, as shown in Fig. 11, is uniform and the stabilization can be considered as poor. The stabilization efficiency of zinc is 89.085%, the lowest among the elements spiked into the samples. Given its uniform distribution and increase in leach rates when fly ash was added to the sample (Table 9), it could be hypothesized that zinc is potentially stabilized chemically since its leach rate increased dramatically from 29.885 mg/L to 818.611 mg/L with the addition of fly ash.

Mercury could be largely stabilized physically since its leach rate increased from 393.682 mg/L to 471.85 mg/L. Fly ash does not contain sufficient mercury to cause such an increase in leach rates. The view of the distribution of mercury is limited (Fig. 11) although it still appears to be localized in the fly ash-geopolymer. Localized distribution suggests that the mercury is present as a precipitate in the fly ash-geopolymer system and the mechanism of stabilization is likely to be physical, where the precipitated mercury compound is encapsulated by the geopolymer. This agrees with leach rate data, particularly where samples with and without fly ash are compared.

The stabilization mechanism of lead is difficult to speculate since the leach rates are similar for samples with and without fly ash. In Fig. 11, the distribution of lead and zinc is almost homogenous across the mapped region, lead and zinc, which have similar distributions, do not have similar leach rates. The stabilization of lead can be considered as good and zinc can be considered as poor. Other factors such as the ability of the heavy metal to bind with aluminum in charge-balancing roles, the solubility of the compounds formed during synthesis, other bonds (covalent) formed with geopolymer reactants, and porosity of the geopolymer all effect the stabilization of the heavy metals. However, lead is an unlikely element to be stabilized in charge-balancing roles[44]; therefore, other mechanisms are responsible. Since the distribution of lead is uniform and the geopolymer is highly porous, stabilization could occur through other types of bonding; for example, covalent bonding to hydroxide groups, although not particularly obvious from these results.

It appears that barium could interact with fly ash to reduce the leach rate of barium. This would explain the successful stabilization of barium in fly ash samples. The stabilization of barium is good regardless of the mass of barium in the sample and the presence of fly ash, although lower leach rates were obtained when Fly ash was present in comparison to stabilization by pure geopolymer. Since the distribution is relatively localized in small regions, as shown in Fig. 11, it can be hypothesized that barium mostly forms precipitates during geopolymer synthesis and, therefore, its leaching will remain low regardless of the compounds in the fly ash-geopolymer paste.

SEM also was used to observe the microstructure of the geopolymer stabilized fly ash samples as well as the interaction of the fly ash particles with the geopolymer phase. As for the heavy metal spiked samples, electron dispersive x-ray analysis was used to determine compositions of regions, area, and lines in fly ash-geopolymer samples. Samples ranging from pure geopolymer to 60–80 wt% fly ash-geopolymers with $Si/Al = 2$ were analyzed by SEM and EDX to give a complete picture of the various stages of interaction based on the percentage of fly ash in the sample.

Addition of small amounts of fly ash results in the dissolution of fly ash particles and the distribution of fly ash constituents throughout the geopolymer network (Fig. 12). Back-scattered electron BSE imaging better demonstrates the distribution by showing elements with greater atomic mass as lighter regions on the image.

Increasing the percentage of fly ash to 40 wt% alters the appearance of the BSE image (Fig. 13). The geopolymer phase is less dominant and fly ash particles appear to be undissolved. At this percentage, there is insufficient geopolymer to encapsulate the fly ash particles and binding into a consolidated mass occurs on a surface reaction basis. Therefore, samples with 40 wt% fly ash and 60 wt% geopolymer exhibited an increase in leaching that can be related to the observations in the SEM images.

At 60 wt% fly ash, the BSE image shows the geopolymer phase as small dark regions distributed throughout the lighter fly ash phase (Fig. 14). A difference in structure between the fly ash and geopolymer can be seen when comparing the dark

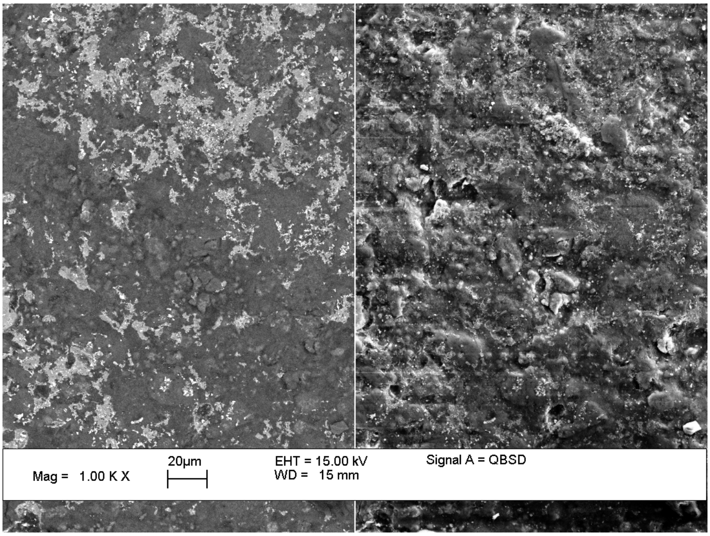


Figure 12. 20 wt% Fly Ash and 80 wt% Geopolymer.

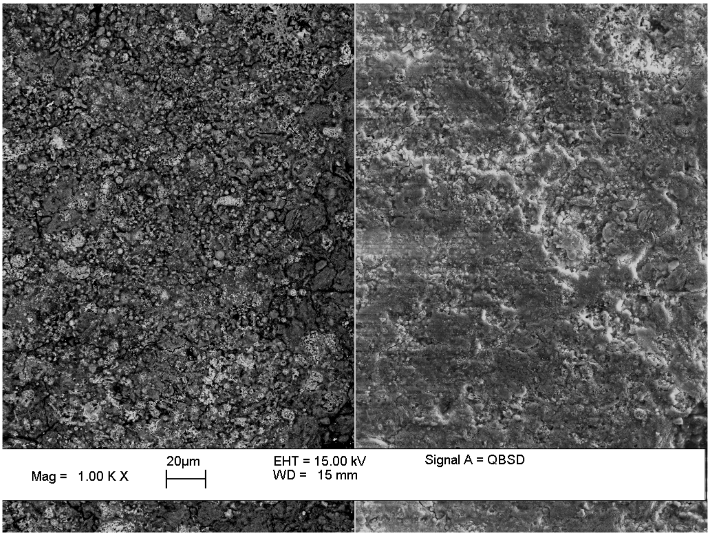


Figure 13. 40 wt% Fly Ash and 60 wt% Geopolymer.

regions on the BSE image with the SE image. Similar geopolymeric features have been observed in other samples.

A sample of 20 wt% fly ash-geopolymer with Si/Al = 2 was mapped using EDX (Fig. 15). The map shows an ash particle in the middle of the image that has two compounds. The outer ring consists of calcium and magnesium compounds and the inner

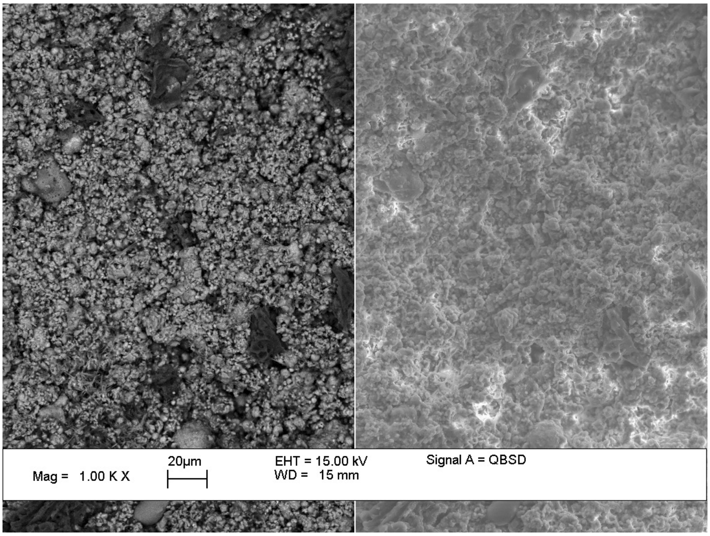
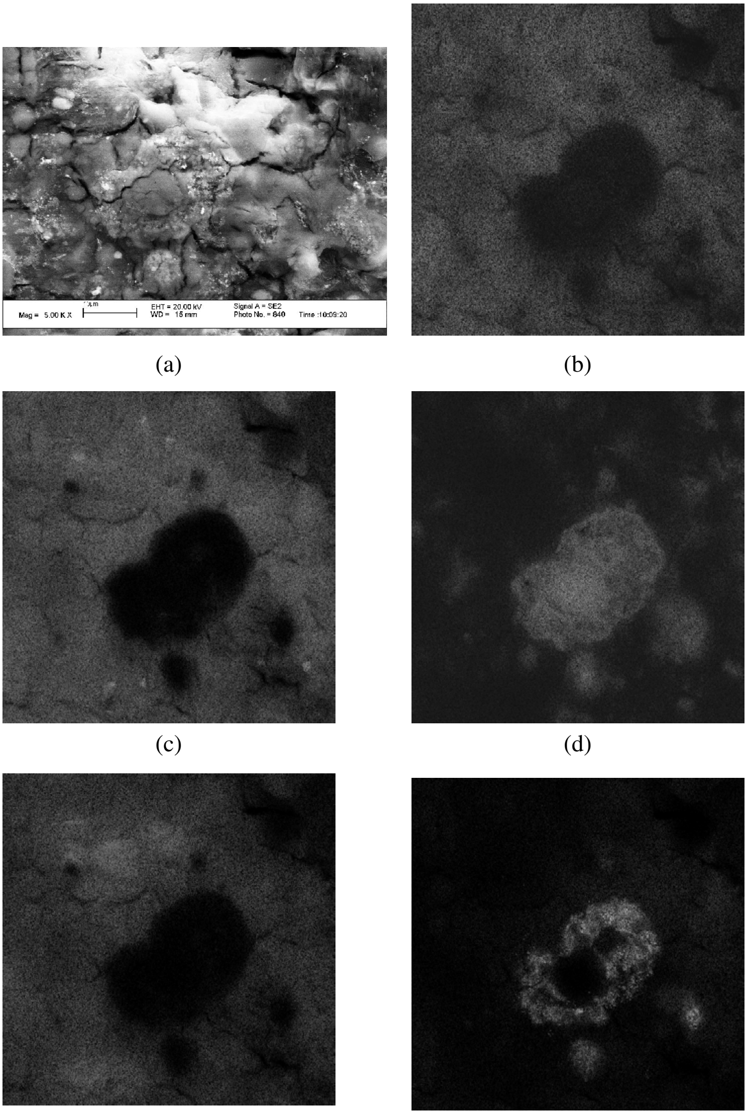


Figure 14. 60 wt% Fly Ash and 20 wt% Geopolymer.

circle contains mainly calcium. The maps appear similar in terms of distribution of silicon, aluminum, and sodium with only calcium being observed as uniformly distributed. Calcium can participate in geopolymeric reactions; therefore, it is possible that some calcium is bound chemically in the geopolymer with charge balancing roles. Magnesium is retained in the ash particle even after being exposed to high pH and the mixing action of geopolymerization. However, there is an area around the particle where magnesium has leached into the geopolymer.

The uniform distribution of silicon and aluminum implies that a geopolymeric material has formed around the ash particle. Other elements could also have leached from the ash particle that cannot react with the geopolymer and although appear not to hinder its formation, MAS-NMR has shown that the addition of fly ash to geopolymer produces a more disordered structure.

A line profile was conducted by scanning a line and determining its composition (Fig. 16). The results are standardless, which means that only a normalized result could be obtained. The disadvantage is that all compounds are expressed as oxides and an average composition is calculated by the software. Compounds present in low concentrations, when included in the calculation, and compounds in high concentration, when omitted, can introduce a large error. The error was minimized by qualitatively determining which compounds were present and then calculating the composition. Any concentrations that were below 1% were omitted since the hardware cannot accurately determine concentration below this figure. For determining where elements diffuse during geopolymerization, qualitative results are sufficient.



(a) (b)

(c) (d)

(e) (f)

Figure 15. Elemental Map of Particle in 20 wt% Fly Ash Sample. (a) SEM Image, (b) Silicon, (c) Aluminium, (d) Sodium, (e) Calcium, and (f) Magnesium.

The transfer of elements between the fly ash and geopolymer phase is a result of partial dissolution from the synthesis of the matrix. Elements are expected to leach from the fly ash under these conditions just as they are during leaching tests. Figure 16 shows the same particle embedded in the geopolymer. The line scan shows a high concentration of calcium in the center of the particle (section A) that was initially observed in Fig. 15. The outer ring (section B) shows a high concentration

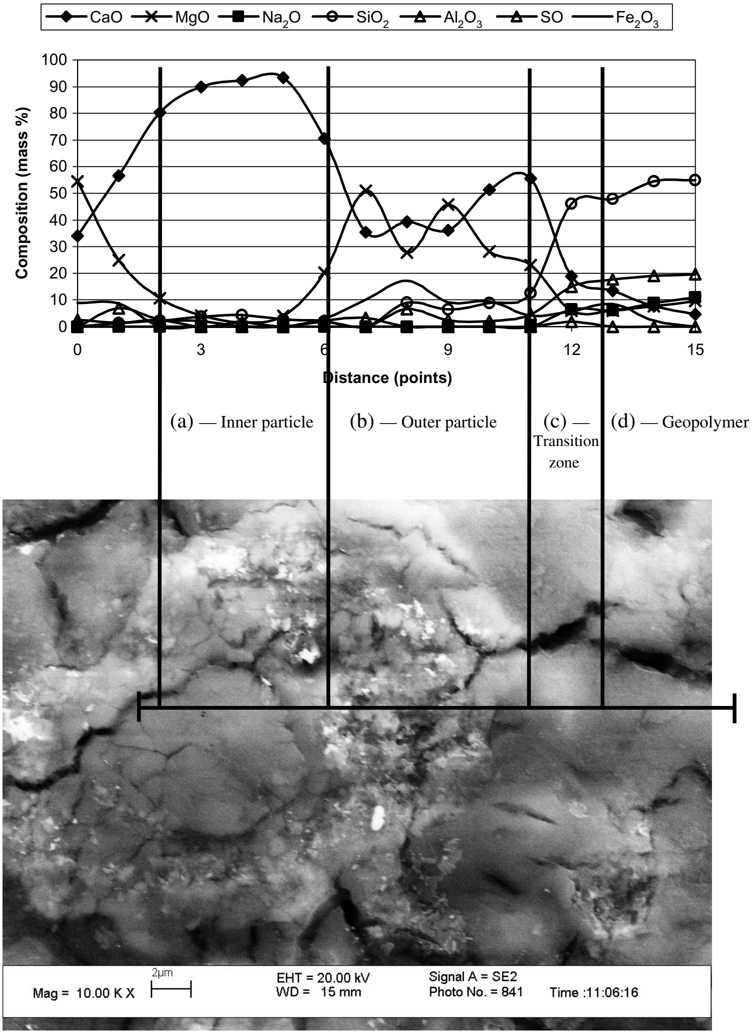


Figure 16. Line Profile of Ash Particle Embedded in Geopolymer.

of both calcium and magnesium. Section D shows increased concentrations in silicon, aluminum, and sodium, the ratio for which is typical of a geopolymer.

Generally, geopolymerized fly ash particles appear to be physically trapped in the geopolymer. However, this is not the case since a transition zone exists between the ash particle and the geopolymer. The transition zone can be seen in section C of Fig. 16 and is approximately 2 μm wide. The concentration of calcium and magnesium decreases and silicon and aluminum increases in this region. Surface reactions are possible where silica, alumina, and calcium react to form complex calcium aluminosilicates as well as geopolymer.

The ratio Si/Na in pure geopolymer is approximately 6 and Al/Na is approximately 2.5. By calculating the ratios in the transition zone, similar results are obtained with Si/Na ≈ 5 and Al/Na ≈ 2.5. These values indicate that the geopolymer formation is theoretically possible even as the transition zone approaches the ash particle. With the ratio of Si to Al remaining constant even as the particle surface is crossed, the transition zone, in terms of the geopolymer, can be made even wider. However, the sodium concentration inside the particle is low; therefore, geopolymer formation could only take place if calcium is in the charge-balancing role. This is unlikely to occur since the formation of calcium aluminosilicates and calcium silicate hydroxides is more spontaneous under uncontrolled conditions such as these. If the surface reaction products are insoluble, then they would reduce the leachable calcium in the sample and provide a physical barrier to prevent further leaching of other elements from the particle.

The examination of SEM images of samples with extra water did not yield any new information. The mechanisms of particle stabilization, encapsulation, and binding are similar to those observed in fly ash stabilized by geopolymers with Si/Al $= 2$. As previously discussed, certain features within each fly ash percentage were also observed.

Fly ash-geopolymer samples with water (Formulation 2) can be seen in Figs. 17–20 and these figures show the change in structure of the geopolymer, as more fly ash is added. The addition of extra water has reduced the fly ash content of these samples and the images reflect this. It can be seen in Figs. 17–20 that the texture associated with the geopolymer phase is still present until the addition of

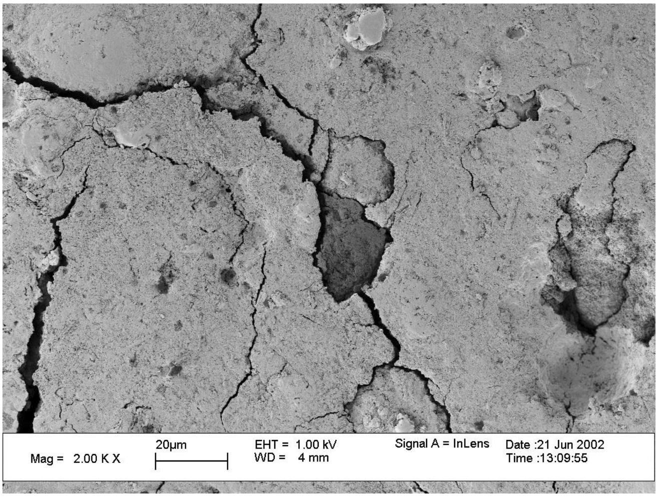


Figure 17. 20 wt% Fly Ash and Water–80 wt% Geopolymer.

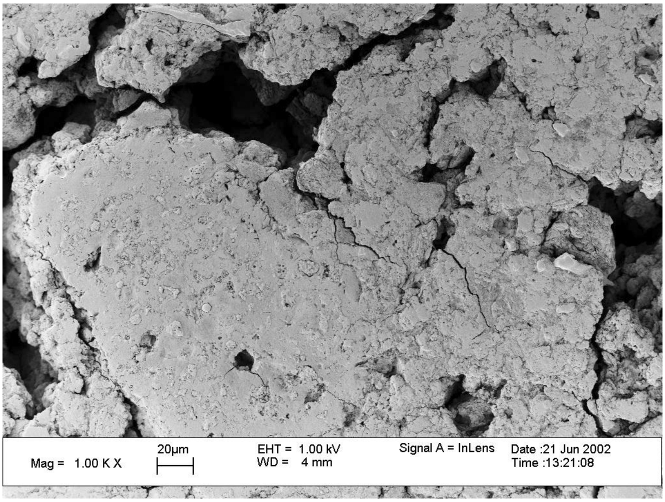


Figure 18. 40 wt% Fly Ash and Water–60 wt% Geopolymer.

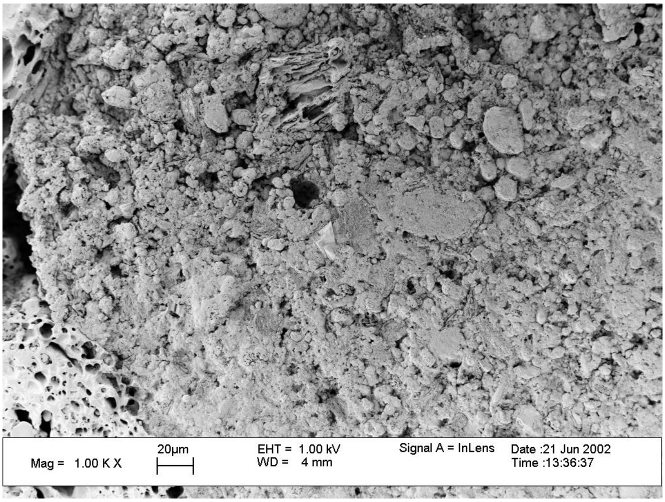


Figure 19. 60 wt% Fly Ash and Water–40 wt% Geopolymer.

80 wt% fly ash and water. Percentages of fly ash and water addition higher than 60 wt% cause the geopolymer to become diluted and the image changes to become dominantly fly ash. At this percentage, the fly ash content is above 40 wt% and the images appear similar to previous samples of 40 wt% stabilized by geopolymers with Si/Al = 2.

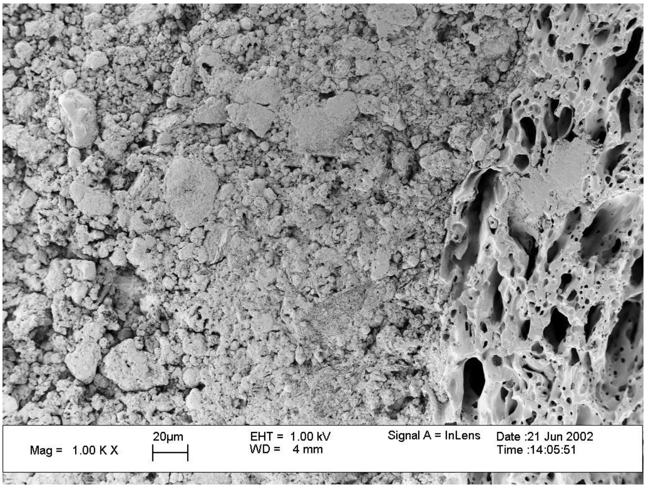


Figure 20. 80 wt% Fly Ash and Water–20 wt% Geopolymer.

7. Discussions of Geopolymer Fly Ash Treatment Process

XRD was used to determine the crystalline phase content and compare the results across fly ash-geopolymer samples with varying percentages of fly ash. XRD spectrums of two fly ash-geopolymer samples are shown in Fig. 21. The low fly ash sample shows the typical amorphous curve associated with geopolymer materials.

The increase in fly ash addition shows that the amorphous curve becomes less noticeable. The cause of this is the reduction in amorphous material, mainly metakaolin, and does not reflect the amount of geopolymer present in the sample. The reason being that metakaolin could remain unreacted and still be detected as amorphous material. In the sample with 20 wt% fly ash (Fig. 21), calcium oxide was not detected, but could have been obscured by the amorphous phases in the sample. In this sample, the three main crystalline mineral phases detected were quartz, sodium silicate, and magnesium oxide. Either the percentage of these calcium compounds has fallen below the detection limit or they have been consumed by reactions with the geopolymer phase or other fly ash constituents. Calcium compounds are observed again when larger percentages of fly ash are added to the geopolymer and analyzed. From this data alone, it is difficult to determine the fate of calcium. However, since fly ash contains 30.6 wt% calcium, there is still approximately 6 wt% in the 20 wt% fly ash sample analyzed in Fig. 21 and is within detection limits of the XRD. Therefore, it is likely that calcium compounds have been reacted rather than not detected.

Figure 4, of untreated fly ash, and Fig. 21 (bottom), of treated fly ash, appear similar suggesting that a large percentage of the fly ash in this sample has not been

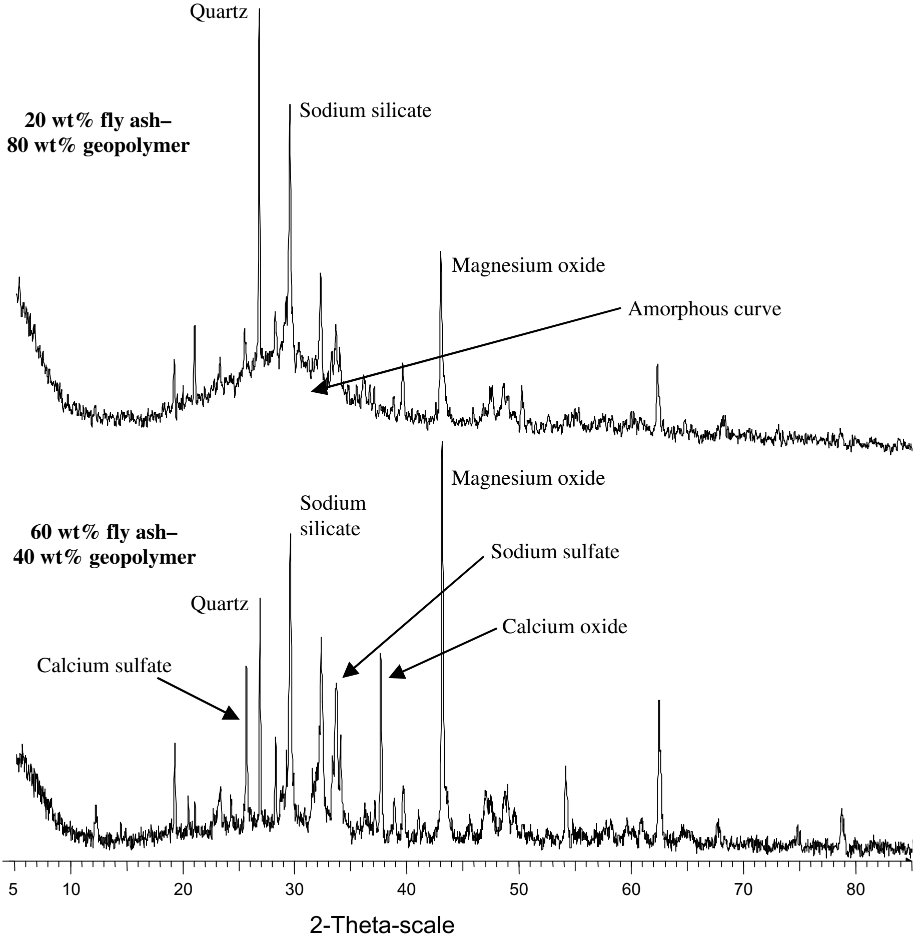


Figure 21. XRD Spectrum of 20 wt% Fly Ash–80 wt% Geopolymer Material and 60 wt% Fly Ash–40 wt% Geopolymer Material.

affected by the geopolymer. This is to be expected since fly ash is the dominant phase and the geopolymer is limited. In terms of water content, less than 32.2 wt% total moisture changes the consistency of the sample and inhibits dissolution and transportation of fly ash constituents. However, magnesium oxide, with a low solubility, was detected in both samples. Regardless of whether or not the sample contained enough liquid to dissolve the fly ash, magnesium oxide was unreacted.

To a certain degree the fly ash-geopolymer system is a mass of competing reactions among sodium silicate, sodium hydroxide, metakaolin, and fly ash. XRD has shown this becomes more apparent, as the percentage of fly ash in the geopolymer is reduced and certain mineral phases are not detected. At higher percentages of fly ash, the amount of possible geopolymer phase is limited; therefore, when the percentage of fly ash is increased, the reactants that are critical to geopolymer formation are diluted and further consumed by other reactions with the fly ash.

Fly ash-geopolymers from formulation 2 with additional water were analyzed using XRD and the results were similar to those obtained from fly ash-geopolymers from formulation 1 (Si/Al = 2). As previously, the samples analyzed contained 20 wt% fly ash and water–80 wt% geopolymer and 60 wt% fly ash and water–40 wt% geopolymer. By comparing Figs. 21 and 22, it can be seen that the amorphous curve that is typical of geopolymeric materials is present in both samples with 80 wt% geopolymer. With increased addition of fly ash, the amorphous curve is reduced as seen in samples with 40 wt% geopolymer. Once again, this is the result of the reduced mass of metakaolin in the sample with 60 wt% fly ash and water, as in fly ash-geopolymer samples with Si/Al = 2.

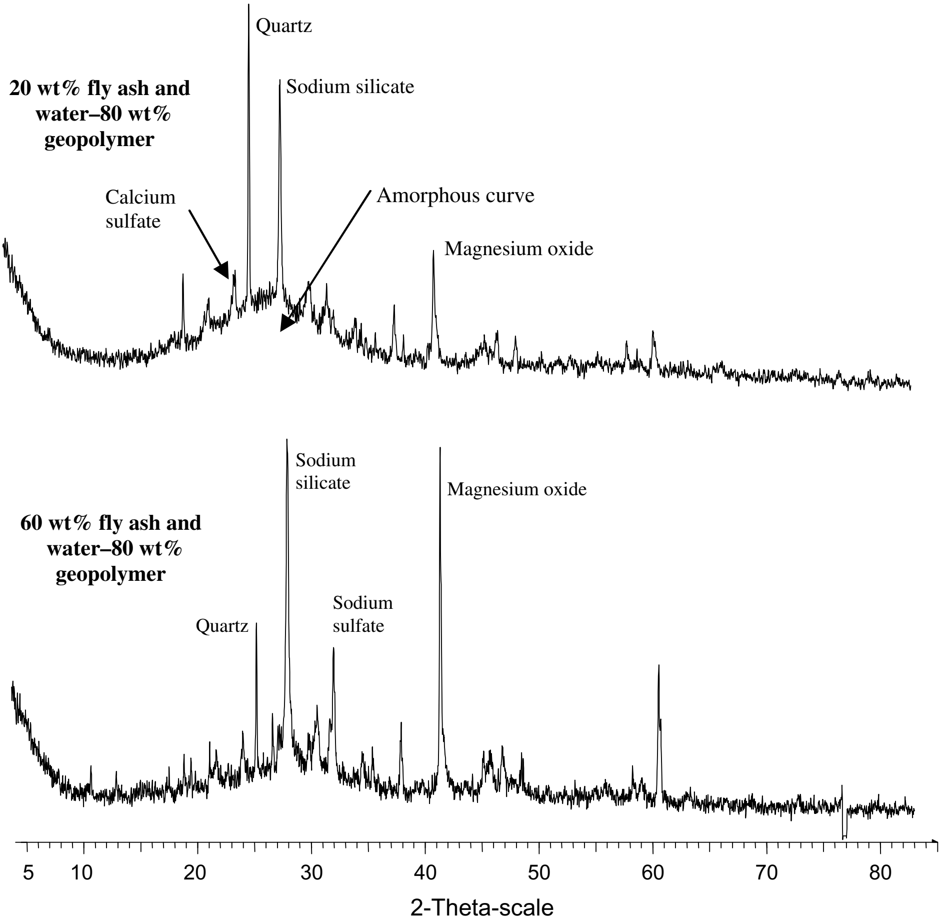


Figure 22. XRD Pattern for 20 wt% Fly Ash and Water–80 wt% Geopolymer Material and 60 wt% Fly Ash and Water–40 wt% Geopolymer Material.

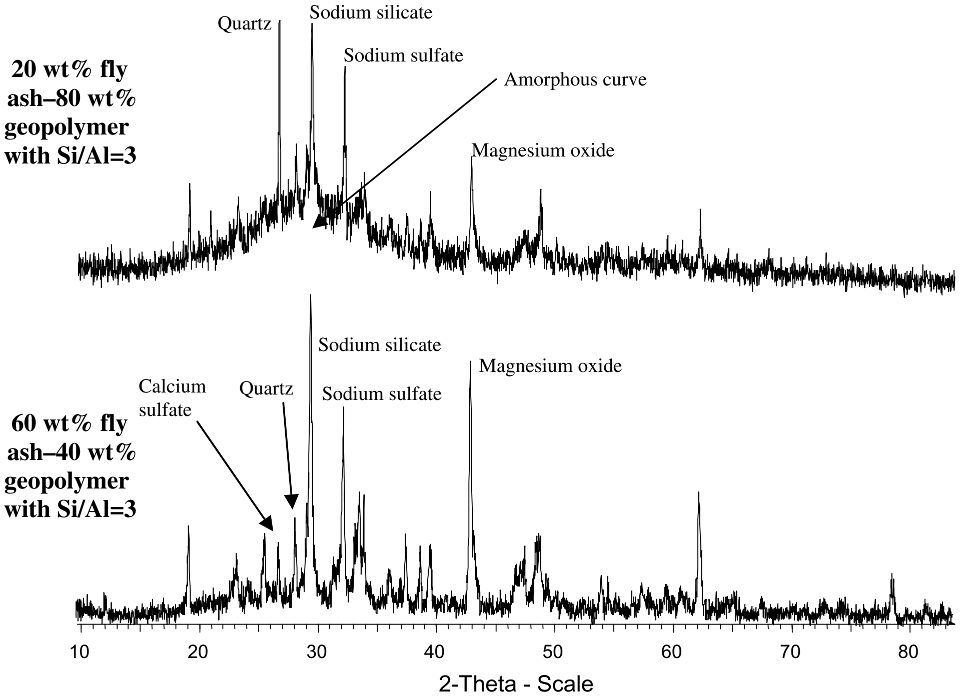


Figure 23. XRD Spectrum of 20 wt% Fly Ash–80 wt% Geopolymer and 60 wt% Fly Ash–40 wt% Geopolymer with Silica to Alumina Ratio of Three.

Similar mineral content was also observed in Fig. 22 (sample with 60 wt% fly ash) with the major mineral phases detected being again, quartz, sodium silicate, and magnesium oxide. In this sample, however, sodium sulfate was detected instead of calcium sulfate. The major mineral phases present in 20 wt% fly ash–80 wt% geopolymer with silica to alumina ratio of three (formulation 4) (Fig. 23) are quartz, sodium silicate, and magnesium oxide. In previous analysis, calcium oxide and calcium sulfate were also observed which are not detected here.

Mineral phases present in 60 wt% fly ash–40 wt% geopolymer are calcium sulfate, quartz, sodium sulfate, magnesium oxide, and sodium silicate (Fig. 23(b)). The absence of an amorphous phase indicates that perhaps there is not enough clay to be detected or most of the metakaolin is dissolved and then consumed by side reactions with fly ash. As mentioned previously, there were no new mineral phases created with the increased addition of silica. The increased addition of silica did not promote the formation of insoluble silicates as hypothesized and suspected from leaching results.

Comparing all the XRD data presented thus far, a trend emerges in the detection of calcium sulfate. The high water content in samples with extra added water dissolved calcium sulfate and calcium was liberated. The sulfate could react with sodium from sodium hydroxide and produce sodium sulfate. This would explain why sodium sulfate was not detected in raw fly ash but was detected even as a minor phase in geopolymerized samples. The fate of calcium would lie in the compounds it forms and charge balancing roles.

8. Potential uses of Fly Ash-Geopolymer Materials

Compressive strength tests were conducted on samples to determine the potential applications, if any, of the stabilized material and to determine whether or not the material will withstand general handling in a manufacturing process. Since fly ash is considered as a raw material, the manufacture of value-added products can reduce the cost of a stabilizations process by cost recovery and also create new materials with perhaps specific applications. This kind of practice not only reduces the costs of the stabilization process but also reduces the mass of fly ash that must be stabilized. The compressive strength of these materials, with further research, could be sufficient to manufacture bricks, pavers, or lay garden paths and driveways or other domestic applications where strength is not critical. In this study, two to four samples were used to generate the compressive strength data, less than the five to six as recommended by the testing procedure, but the reduced sample size was enough to determine the potential applications of the material. If the material is considered for use in building applications, then more thorough testing will be required.

The compressive strength curves for all formulations are shown in Fig. 24. The compressive strength of the pure geopolymer with $\mathrm{Si/Al} = 2$ at zero fly ash addition is low, less than 2 MPa. The addition of small percentages of fly ash increases the strength of the geopolymer considerably. The maximum observed for fly ash-geopolymers from formulation 1 was approximately 18.5 MPa for 20 wt% fly ash addition. The maximum obtained for fly ash-geopolymers from formulation 2 was 14 MPa. It was found that the compressive strength of these samples was generally lower than those of samples of geopolymer with $\mathrm{Si/Al} = 2$. This can be explained by extra water added to the fly ash. Once the fly ash, water phase, and the geopolymer phase have been blended together, the dissolved geopolymer precursors are diluted, making polymerization and network formation difficult and leading to poorer compressive strength. These values of compressive strength are still acceptable for nonload-bearing applications such as pavers and garden edging. The addition of sodium hydroxide increased the overall compressive strength of the fly ash-geopolymers (Fig. 24). The strength maximum observed for fly ash-geopolymers was 27 MPa, which is 8.5 MPa greater than the fly ash-geopolymer with $\mathrm{Si/Al} = 2$ and 13 MPa greater than fly ash-geopolymers with extra water. However, in other formulation, the maximum compressive strength for fly ash-geopolymers

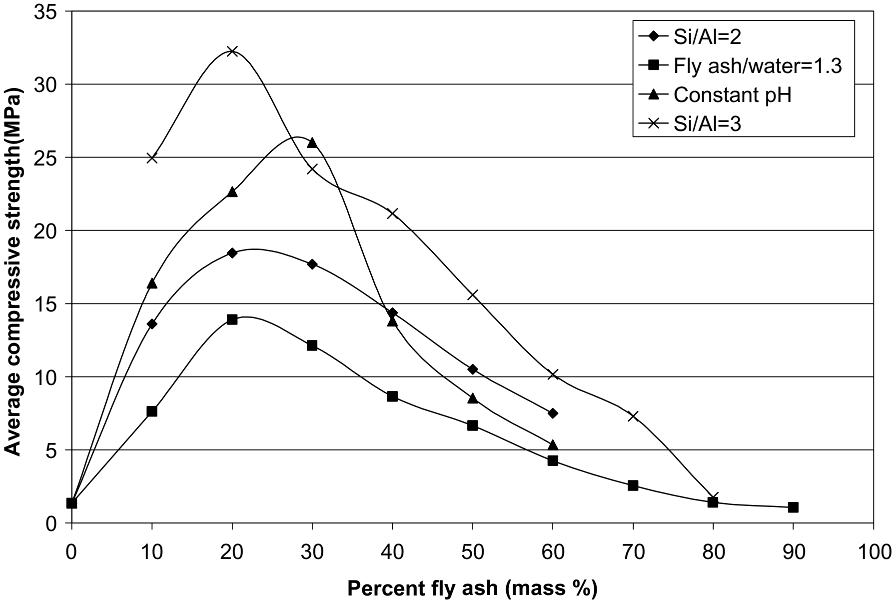


Figure 24. Compressive Strengths of Fly Ash-Geopolymers.

was observed at 20 wt% fly ash addition and for this series, the maximum occurs at 30 wt%. Since more sodium hydroxide is added with each increase in fly ash addition, the compressive strength will increase until the fly ash interferes with network formation. In this case, this occurs at 30 wt% fly ash, whereas in previous formulations, this occurred at a lower percentage since less sodium hydroxide means poorer network formation. The maximum achieved for formulation 4 (Si/Al = 3) was approximately 33 MPa for 20 wt% fly ash addition. Comparing this formulation to the other formulations tested, the compressive strength is greater.

The trends shows that strength increased with increasing addition of fly ash until a maximum is reached, then a decrease in observed. The decrease in strength can be explained by excess fly ash in the geopolymer network, as observed by SEM images. Fly ash particles only partially dissolve during synthesis and act as a filler material. As the percentage of fly ash is increased, a point is reached where there is not enough geopolymer material to bind the fly ash and strength is reduced.

However, the processing of the uncured materials must also be considered. The compressive strength of fly ash-geopolymers is highest at 10–40 wt% and experiments have shown that curing times decrease dramatically to several minutes at 40 wt%. The fast curing time would be impractical in manufacturing as the material could not be removed from the mixing vessel in time. At an addition of 20–30 wt% fly ash, the three criteria of compressive strength, leaching, and processing can be met without having to sacrifice any particular property.

9. Summary

Fly ash had been a problem for many power stations. Several methods of stabilization and treatment are currently available to reduce the leach rates of elements, especially heavy metals, from fly ash and reduce the environmental impact after disposal. This study has focused on using a relatively new encapsulation technique by stabilizing the fly ash using a geopolymer. Literature has shown that geopolymers have desirable properties that make them suitable for waste encapsulation for a variety of wastes. The potential of a geopolymer to stabilize fly ash was, therefore, explored through the use of four geopolymer formulations.

Each formulation addressed fundamental properties of the geopolymer and the mix design was altered to attempt to maximize the stabilization. Initially, a geopolymer with a molar silica to alumina ratio of two was employed that has been proven in literature to be effective at the stabilization of heavy metal. The formulation was then altered by increasing the water, sodium hydroxide, and finally the silica contents to determine the effect of stabilization. The fly ash was added to the geopolymer with increasing percentages to determine the maximum capacity of the fly ash in the geopolymer.

Preliminary analysis of the leach rates of untreated fly ash helped to target a list of seven elements including arsenic, barium, beryllium, selenium, strontium, calcium, and potassium. The lowest leach rates were observed using the geopolymer formulation with $Si/Al = 2$, although the other three formulations also had good results. The geopolymer with $Si/Al = 2$ reduced the leach rates of strontium from 31200 μg/L in unstabilized fly ash to 1100 μg/L at 10 wt% fly ash addition (96% reduction in leaching compared to unstabilized fly ash), with a clear trend indicating the stabilization had been successful. Selenium, which is also present in relatively high concentrations, was stabilized with a leach rate reduction from 740 μg/L in unstabilized fly ash to 60 μg/L at 10 wt% fly ash (92% reduction). Arsenic, barium, and beryllium also showed excellent stabilization results with significant reductions in leach rates.

To understand the stabilization of heavy metals, which are present in low concentrations in this particular fly ash, a series of spiked samples was synthesized with increased percentages of heavy metals compared to the fly ash. The heavy metals chosen were lead, mercury, zinc, and barium. The geopolymer was very effective at reducing the leach rates of these heavy metals and encapsulation efficiencies greater than 99% were achieved for most of the samples, even with the addition of small percentages of fly ash.

SEM was used to determine the fly ash particles–geopolymer phase interactions and it was shown that a transition zone exists between the particle and the geopolymer phase. As more fly ash is added to the geopolymer paste, the mechanism of interaction change from complete fly ash dissolution in the geopolymer phase to surface reactions between the fly ash and geopolymer to consolidation by other mechanisms not associated with geopolymer stabilization. At fly ash percentages of less than 30 wt%, the mechanism for particle stabilization is particle dissolution and complete encapsulation by the geopolymer phase. Between 40 wt% and 50 wt%, the mechanism is particle surface reactions and around 60 wt% the mechanisms for stabilization and consolidation are based on the ability of calcium sulfate to form hydrates. This was the case for all formulations and it was found that alterations to the geopolymer chemistry did not change the mechanisms of interaction.

The maximum addition of fly ash to the geopolymer with Si/Al = 2 was 60 wt%. The addition of water decreased the maximum addition to 50.9 wt%, but this was the result of including water in the formulation and replacing a portion of the fly ash. Increasing the sodium hydroxide content did not have any effect on the maximum addition and 60 wt% was achieved. Changing the silica/alumina ratio to three increased the maximum addition to 80 wt% fly ash, which was possible due to the extra water added with the sodium silicate solution. Overall each formulation showed maximum allowable additions of over 50 wt% fly ash, although sample integrity was compromised in some cases.

When tested for compressive strength fly ash-geopolymer with Si/Al = 2, showed good results. The maximum strength achieved for this formulation was 18.5 MPa. Addition of water to the fly ash and geopolymer reduced the maximum compressive strength to 14 MPa by diluting the geopolymer reactants. Increasing the sodium hydroxide content increased the compressive strength of the fly ash-geopolymers to 27 MPa. This was attributed to the higher pH of the activator solution. Increasing the silica content also increased the maximum compressive strength to 33 MPa. This was the highest compressive strength achieved throughout all formulations.

The high compressive strength of the fly ash-geopolymers showed that these materials can be considered for some potential applications that included domestic applications such as pavers, garden bricks, garden edging, paths, driveways, and even statues or other art products. With further research into compressive strength development, these materials could be used in applications where compressive strength is more important. However, considering the good stabilization and low leach rates of the fly ash once it has been encapsulated in the geopolymer, these materials can be currently sold to recover costs of stabilization and reduce the volume of fly ash that needs to be stabilized.

In terms of stabilization, which is the most important property of synthesizing fly ash-geopolymers, the geopolymer formulation with Si/AL = 2 showed the best results. However, mechanical properties were best in geopolymers with silica to alumina ratios of three.

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Chapter 9

MUNICIPAL SOLID WASTE RECOVERY AND RECYCLING

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Abstract

Recycling is a series of activities that includes collecting recyclable materials that would otherwise be considered waste, sorting, and processing recyclables into raw materials, and manufacturing them into new products. In this chapter, the collection of recyclable materials in both developed and developing countries are reviewed. The physical and chemical recycling methods are also included with necessary engineering calculation. Finally, the marketing of recyclable materials is also depicted.

Keywords: Waste recycling, waste recovery, recyclable material, waster separation, material recovery facility.

1. Introduction

Solid waste is a serious environmental problem facing many of the urban and industrial areas of many countries. Substantial progress has been made in the past decade to improve waste management practiced in many regions. In particular, there is a large untapped potential in recycling, waste collection and treatment, and disposal systems are only just beginning to take shape in most areas of developing countries.

Recycling is a series of activities that includes collecting recyclable materials that would otherwise be considered waste, sorting, and processing recyclables into raw materials, and manufacturing them into new products. Commonly, recycled materials include:

- Paper — newspaper, office paper, cardboard, and other paper types
- Glass — bottles and jars (clear, green, and amber)
- Aluminum — beverage containers
- Other metals — steel cans, auto bodies, refrigerators, stoves, and batteries
- Plastics — soda bottles, milk jugs, bags, and detergent containers
- Yard trimmings — grass, leaves, and shrub and tree clippings
- Used motor oils — vehicle crankcase oil

In general, the recycling process includes of collecting, separating and processing, and marketing recycled products. Collecting recyclables varies from community to community. Four primary collection programs widely used in many countries are curbside collection, drop-off centers, buy-back centers, and deposit/refund programs. After collection, recyclables are sent to a materials recovery facility to be sorted and prepared into marketable merchandises for manufacturing. Then, recyclables are sold and bought just like any other products and prices for the materials change and fluctuate with the market. Recycled materials also used in innovative applications such as recovered glass in roadway asphalt or recovered plastic in carpeting. As consumers demand more environmentally sound products, manufacturers will continue to meet that demand by producing high-quality recycled products.

In general, U.S. EPA has identified four basic solid waste management options, which are source reduction, recycling and composting, combustion, and landfills. Among all alternatives, recycling is believed to be the most environmentally benign. The many benefits from recycling are as follow[1]:

- Recycling helps sustain the environment for future generations.
- Recycling conserves natural resources such as water, minerals, and timbers.
- Recycling decreases emissions of greenhouse gases that contribute to global climate change.
- Recycling prevents pollution caused by the manufacturing of products from virgin energy.
- Recycling saves energy.
- Recycling reduces the need for landfilling and incineration.

Recycling is perhaps the most positively perceived and doable of all the waste management practices. It is possibly the most practical way to reduce solid waste, boost the economy, and conserve natural resources.

Recycling is a cost-effective method of solid waste management. Within eight years of the introduction of a curbside recycling program in Madison, Wisconsin, the net annual cost of solid waste services decreased from \$158 per household to \$139. A recycling rate of 49% reduced the number of garbage routes needed and helped hold landfill tipping fees in check. In a similar fashion, a net annual cost-saving of more than \$650,000 resulted from the integration of curbside recycling into the solid waste management system in Mesa, Arizona.[1]

Besides economic benefits, recycling also provides environmental benefits. A recent analysis of several studies concluded that the environmental impacts of recycled-content products are less than those of virgin products when the two are compared over their entire life cycles.[2] Studies show that less energy is used to manufacture products from recyclables than from virgin materials.[3] Moreover, recycling helps reduce greenhouse gas emissions by decreasing the energy needed to make products from virgin materials (reducing the burning of fossil fuels) and thereby reducing emissions from incinerators and landfills, which are the major sources of methane.

2. Collection of Recyclable Materials

Collection of recyclable materials in developed countries normally includes curbside collection, drop-off centers, and buy-back centers; while in developing countries, it includes formal and informal collections.[4–6] The former is the recyclable collection system that is operated by the government or the recycling business and the latter is the waste collecting by waste pickers from waste containers or disposal sites. In some countries, the collection system is a combination of these applicable methods.[5]

2.1. *Collection of Recyclable Materials in Developed Countries*

2.1.1. *Curbside Collection*

Curbside collection is the most convenient way for a household to recycle. It is widely used in many countries. This system offers scheduled pickups of separated, recyclable products from the curb. Within most communities, solid waste from single-family residences, multi-family units, and businesses are typically collected at the curb in various-sized plastic or metal containers. Collection includes three basic steps: (i) source separation, (ii) collection in one of a variety of collection vehicles, and (iii) hauling to a processing destination or final market. In this system, the residents typically have responsibilities for separating the recyclable materials from the waste stream and placing them at the curb for a regularly scheduled pickup. Many types of curbside collection have been used in many countries in different programs:

2.1.1.1. Source separation

In this type of collection system, residents are most often asked to keep a number of recyclables separated from the mixed waste by storing them in centrally located bins. The residents subsequently place these bins at the curb for pickup. This requires that building management must provide storage space for centrally located bins. The materials may or may not require subsequent processing to meet market specifications. The advantage of this system is that a high material quality of recyclables is generally achieved. However, specially designed multi-compartment recycling vehicles must be used. The collection approach is labor intensive.

2.1.1.2. Curbsort source separation

This alternative method offers residents the convenience of being able to drop all recyclable into a single bin at the curb. The recyclables are sorted at the curb, by the collector, into the compartments of the sorter vehicle. Owing to the quality control function that the curbside sorters perform by rejecting nonrecyclables, this system is able to achieve a high material quality rate. However, curbsorting is much more labor intensive and there is an upper limit to the number of homes from which can be economically collected as the size of the route increases.

2.1.1.3. Commingled source separation

This system offers residents to place recyclables into a single bin, which is separated from nonrecyclables. These bins are set out at the curbside for commingled collection. Newspapers are often kept separate from the rest of the commingled recyclables to prevent contamination. The commingled recyclables are then subsequently separated at a material recovery facility (MRF). Those recyclables are segregated into each recyclable component such as glass, aluminum cans, plastic bottles, etc.

2.1.2. *Drop-off Centers*

The drop-off center is one of the types of permanent recyclable collection centers. This center requires residents to bring their recyclable materials to the site and place the burden of separation on the provided containers. This center serves as a good starting point for a community to use in implementing a comprehensive recycling program. Drop-off centers can be designed to collect a wide range of material, including aluminum, glass, newspaper, old corrugated containers, high-grade paper, and various types of metal. Drop-off centers are normally located on free properties, such as schools, church properties, grocery stores, and sometimes shopping center parking lots. Various sizes of drop-off centers can be set up; for example, it can be as simple as a few collection bins in small area or as complex as a large processing center located at disposal sites. The complex drop-off centers require additional equipment for volume reduction or improvement of recyclable quality for marketing purpose. Basic tools such as weigh scales and storage containers are necessary. Additional processing equipment for the complex drop-off centers might be conveyors, shredders, magnetic separators, glass crushers, and balers. Some of the many factors affecting the selection of drop-off center types include waste composition, types of material collected, amount of recyclables, marketability, site maintenance, and access to the centers.

Drop-off centers are suitable both in urban and suburban areas to serve multi-family residences and less densely populated single-family districts. These permanent collection centers may be the most practical choice in the areas where disposal is cheap and in some areas where there is a long driving distance between residences. To achieve high collection efficiency, drop-off centers need to be in convenient locations, within five miles of the home.[7] These centers can be employed for collecting hazardous wastes such as batteries, used oil, bulbs, and other household hazardous wastes.

2.1.3. *Buy-back Centers*

The buy-back center is another type of permanent recycling center. It is much more similar to a recycling business than a drop-off center in that the recyclable dealers provide economic motivation by paying cash for recyclables. With this economic incentive, residents may be willing to travel further to a buy-back center than drop-off centers. Generally, the buy-back centers have a larger service area than drop-off centers. Sizes of buy-back centers can vary considerably depending on the participation of people and types of waste that is required by the dealers. These complex centers will purchase a wide variety of recyclables reflecting the need of processing equipment for these materials. In general, establishing a complex buy-back center is more costly than a drop-off center because of the additional equipment and labor needs. Equipment must be matched to the properties of the materials to be processed, to the throughputs required, and to the desired specifications of the product. Space limitation is another factor that has to be considered for equipment selection for buy-back centers. Additional equipment for complex buy-back centers are conveyors, shredders, magnetic separators, glass crushers, and balers. In comparison, a simple buy-back center with very limited processing can be combined with scales and a storage area or containers. A number of recyclers and end purchasers have attempted to reduce the costs of constructing and operating buy-back centers by installing automated facilities using reverse vending machines in convenient locations to accept beverage containers such as aluminum can. A reverse vending machine functions as a mini-buy-back center and it can be located outdoors or indoors. They give cash or in-store credit in exchange for the recyclable wastes. Most units are produced to accept aluminum cans, while some of them are created to receive steel cans, plastic bottles, and/or glass bottles. Some of them have shredders to reduce the size of polyethylene terephthalate (PET) or glass bottles.

2.2. *Collection of Recyclable Materials in Developing Country*

2.2.1. *Recycling through the Formal Sector*

In general, both public and private sectors are active in management of solid waste in developing countries. In many countries, the major recycling group is the formal sector, including the government and recycling businesses. The recyclable collection system, known as curbside collection system, is mostly processed by government. The various methods of curbside collection employed in developing countries are the same as those mentioned earlier for developed countries. In developing countries, recycling is more often done by a collection crew. The revenue from recyclable materials belongs to the crews who unofficially separate the waste.

Formal recycling is also done by legal businesses that pay taxes and work under a commercial name. They buy recyclables from individuals who separate materials with commercial value in their homes and sell them to the recycling companies. Commonly, they buy recyclables from itinerant waste pickers. Some businesses collect all types of waste that can be sold for recycling or transformation; however, some buy certain kinds of wastes; for example, some will only buy plastics, others will buy solely paper and cardboard, some others will buy only metal or glass, etc. In all cases, the received wastes are only the recyclables that have a commercial value and can be sold to commercial recyclers or manufacturing businesses.

Recyclers are companies that transform recovered materials without commercial value into new products or raw feedstock for certain industrial processes. The purchasing prices of recyclables vary depending on the offer and demand of the product. This price uncertainty creates a problem about which type of waste to invest time and money in order to make a profit for the recycling system. Another factor affecting the purchasing price and selling opportunity is the quality of the recyclables. Some recyclers avoid buying salvaged materials from individuals or landfill pickers because the items are very dirty. The matter of cleanness makes the materials difficult and sometimes impossible to sell. The recycling process for those objects below quality is more complex and costly.

Some recyclers set up their own businesses as buy-back centers. At these sites, materials are processed that are compacted, shredded, grinded, packed, and sold to the industry for manufacturing processing.

2.2.2. *Recycling through Waste Pickers (Informal Collection)*

An important characteristic of waste recovery and recycling in developing countries is the participation of the informal sector. Various studies reveal that the sector is mainly involved in recovery and resale of most of the recyclables.[4,8,9] The major groups of collectors playing an important role in this system are the waste pickers, persons who literally live on waste collecting and are completely dependent on its sale for their food and clothing.[4] Recyclable wastes become a source of income for waste pickers and their families. They help reduce the volume of waste to be disposed, and also keep materials in circulation for recycling or reuse. The informal collections in this pattern occur in many developed countries due primarily to insufficient municipal services, which create a large need for informal waste collection; as a consequence, it provides an opportunity for income for the poor. Taylor presented the results of some studies done in different countries on the contribution of this informal sector in tasks of solid waste recovery, as shown in Table 1.[9]

In Mexicali, Mexican itinerant waste pickers roam the streets collecting objects (for example, aluminum cans, plastic, and cardboard, among other items). They go from house to house collecting or buying refuse that residents consider of no value and sell it to gathering centers.[4] It has been estimated, for instance, that around 2% of Mexican population live off recycling-related activities, including 30,000 people in the metropolitan area of Mexico City alone. In Columbia, 30,000 people, roughly 1% out of the country's population, are involved in scavenging activities.

The informal waste collection in Thailand is considerably smaller, with an estimated 25,000 people involved in informal recycling. This includes more than 15,000 individual waste collectors, 2,000 waste agents, and 4,000 scavengers who collect waste from disposal sites and transfer stations. These informal waste collectors are subject to dangerous and unsanitary working conditions. This problem rises from the limited living benefits such as medical insurance and pension plans. The waste pickers, especially, have limited job stability and few education and job opportunities.

Table 1. Contribution of Informal Collection of Recyclables Done by Waste Pickers in Developing Countries.

City	Country	Percentage of Contribution in Recycling of the City's Wastes
Jakarta	Indonesia	25
Surabaya	Indonesia	12
Bangkok	Thailand	5
Ho Chi Min City	Vietnam	7

Source: The World Bank.[5,6]

The following initiatives may provide some innovative ideas to help these people:

Columbia — The support of government agencies and local NGOs has greatly contributed to the improvement of working conditions for waste pickers throughout Columbia. Recycling organizations started to appear during the early 1990s and increasingly became small-scale enterprises and regional cooperatives. These organizations have gradually given their members employment benefits such as subsidized health care, paid vacations, and pensions. Membership improves overall social behavior personality through training and social participation in meetings, activities, and community life. Working conditions of recyclers typically improve after joining cooperatives as they are provided stable access to sources of recyclable materials (away from the landfills), as well as equipment and uniforms.

The Philippines — In 1993, the community of scavengers that lives in Metro Manila's Payatas dumpsite established the Payatas Scavengers Association. This initiative has considerably improved the living standards of its members by promoting programs that aim at stabilizing the monthly incomes of the scavengers. These programs include (1) establishment of home-based solid waste microenterprises where recyclable materials are processed and subsequently resold at a profit and (2) creation of a communal savings fund that provides loans for members of limited and uncertain monthly incomes.

The effectiveness of informal waste recovery may be enhanced through appropriate equipment design and facilities for each stage of the collection and disposal process.[10] The useful activities supporting the effectiveness of informal recovery include improving the organizational capacity of informal workers, improving equipment and facilities for the collection and sorting of material, and coordinating municipal waste collection and disposal operations with informal recovery. Formal public sector workers often engage in some form of scavenging activity on the side, and it may be necessary to specify the rights and recovery conditions of both formal and informal workers.

2.2.3. *Garbage Bank*

The use of a garbage bank is an alternative way to encourage recycling activities at the community level through which participants receive goods or money in exchange for their recyclable wastes. Many types of garbage bank can be employed as follow:

School Garbage Bank: This type of garbage bank is typically set up in local schools where students can bring recyclable waste for collection. Students receive either cash or a certain number of reward points in exchange for their waste. The reward points depend on the volume and type of recyclable material presented and can be redeemed to procure goods such as sports equipment, stationary, and other consumer products. Those supplies are funded with profits from the sale of the recyclable materials. The school garbage bank is typically managed by students, with support from teachers and parents.

Community Garbage Bank: This type of garbage bank is organized by communities and municipalities. For example, the garbage bank in Udonthani, Thailand is run by the community. The members of this bank receive shares of proceeds from the sale of all the collectable recyclable waste. In Phichit, Thailand, profits earned from a garbage bank were used to set up a communal convenience store, where goods are sold to members at a low price.

The "Garbage-for-Eggs" Project: This project was initiated in one of Bangkok's poorest residential areas by Klong Toey Environmental Protection Group. Residents were encouraged to collect recyclable materials and to exchange them for eggs. Within six months of the program's start date, the amount of waste in the community was reduced by 161 tons. This project became a recycling model, now operating in more than 23 communities within Bangkok and other provinces.

3. Processing of Recyclable Materials

The processing of recyclable materials from commingled waste is a necessary task in material recovery for direct use and for reproduction of a new product. The aim of this section is to provide a picture of how materials can be transformed to be appropriate recyclables for recycling purposes. In past, the processing of recyclable materials is mainly relied on physical recycling methods. Nowadays, the chemical recycling methods play an important role in recycle material especially for nonmetallic fractions such as plastics for a further used as fuel. In this section, the physical recycling methods, the classical recycling technology, are described and the chemical recycling methods are introduced.

3.1. *Physical Recycling Methods*

Classical methods and equipment used in physical recycling processing are provided in Table 2.

Table 2. Typical Methods and Equipment Used for Recovery Processing.

Recovery Processing	Activity
Materials handling	Transporting and storage of municipal solid waste and recyclables
Manual sorting	Removing items from the waste stream by hand
Size reduction	Reducing size of solid waste and recyclables
Size separation	Separating materials by size and shape characteristics
Densification	Increasing the density of recovered materials by densification or compacting

Source: Tchabanolgous Kreith.[17]

3.1.1. *Materials Handling*

Material handling is the operation unit used for the transport and storage of materials either from municipal solid waste or from recyclable processes. Conveyors are often used to move the materials to each process unit. As the commingled wastes are on the tipping floor, materials that are not suited for processing may be taken out manually or with mechanical equipment. A loader will push the rests that are likely to be the appropriate recyclable materials, into a receiving bin. Conveyors are often used to move the materials vertically and horizontally. Four basic types of conveyors are widely used for material handling: rubber belts, vibratory conveyors, pneumatic conveyors, and screw conveyors.

Rubber belts are the most popular types of conveyors. They are acceptable for less abrasive and less rugged loads. They are also relatively low in cost. Rigid interlocking belts have been successfully used for raw material conveying. However, they have been found to be ineffective for moving unshredded raw materials. Vibrating feeders are material conveyors equipped with a spring- or hinged-mounted bed, which is swung in an eccentric motion causing the material upon the bed to be transported in one direction. A constant flow of materials is possible using a belt conveyor. As commingled waste can absorb a considerable amount of energy, the vibrating conveyors must supply sufficient energy to achieve material movement. Pneumatic conveyors have been used widely for conveying raw bagged waste in hospitals and large buildings. This conveyor requires air to transfer the materials. Positive pressure or a vacuum may be used to operate this system. The last type of the four conveying methods, the screw conveyor, has found limited use in solid waste handling. The major characteristic of this conveyor is a rotating auger in a narrow channel used to push loose materials forward. It is used basically for dry and free-flowing materials.

3.1.2. *Manual Sorting*

Manual sorting is the operation unit in which personnel physically remove items from the waste stream. This sorting operation includes removal of bulky items and hazardous materials, which would obstruct other processes, off an elevated conveyor into large bins located below the conveyor, and sorting material that need not be processed (for example, corrugated cardboard, bundles of newspaper, and large pieces of metal). This method depends on laborers to hand-separate recyclable materials. Generally, sorters are located at stations on one or both sides of the moving conveyor. Mostly, each sorter is responsible for a specific material. The coding and separation functions in hand sorting are simple to define. Each material is visually classified by its properties such as color, reflectivity, and opacity. Sorter skill is an important factor that can affect overall sorting efficiency. Sometimes, operation training is required.

Another factor that plays an important role in sorting efficiency is the conveyor speed. Typically, the conveyor speed can be adjusted or stopped by the sorter. Reducing the speed of the conveyor does not significantly reduce the overall processing rate and may result in a higher sorting efficiency. Manual sorting may take place on the tipping floor as well.

Manual sorting is suitable for small recovery plants but is unsatisfactory for large-scale recovery and utilization for several reasons, which include: (1) low salvage prices that limit the economic attractiveness of such operations; (2) limited degree of separation since a nominal size work force can be concerned only with removing more bulky pieces; and (3) human fallibility.

3.1.3. *Size Reduction*

Size reduction is the operation unit used for the reduction of both commingled solid wastes and recyclable materials. Several types of size reduction equipment are employed to cut, tear, and pulverize commingled waste and recyclable materials. The bound materials are liberated; therefore, they can be separated from each other and do not interfere with the operation in downstream units. Size reduction increases the homogeneity, the bulk density, and the surface area-to-volume ratio. The increased density provides more efficient in transporting of wastes and less landfill space. A major type of size reduction equipment widely used for solid waste is the shredder.

Hammermills are the most commonly employed type of equipment used for solid waste reduction.[11] Hammermills are normally used for coarse size reduction and a wide range of particle size can be obtained from the cutting process. Two basic types are vertical and horizontal hammermills. In both types, solid wastes are fed from the top and flow through the hammermill by gravity. The different sizes of wastes are obtained from the bottom opening in the unit. In horizontal hammermills, the size of the openings in the floor-grate is used in controlling the size of the reduced materials. In vertical hammermills, the size of the reduced materials is restricted by the opening between the lower hammers and the breaker bars on the inner-housing wall. Typically, the horizontal hammermill has a smaller opening than that of the vertical mill. A particle cannot pass through this opening until it is smaller than the opening clearance. Nature of solid waste, the processing rate, and the particle size are major factors affected on size and power required for hammermill. The reduction size of the material obtained from the hammermill is varied and is a function of solid waste composition, moisture content, feed rate, opening size at the grate, and rotor speed. Pictures of vertical and horizontal hammermills are shown in Fig. 1.

Other types of size reduction equipment are rotary grinders, shear shredders, and pulverizers. Rotary grinders are operation units in which rolling star wheels are pinned to a vertical rotor operating at a relatively high rate of speed. The solid wastes

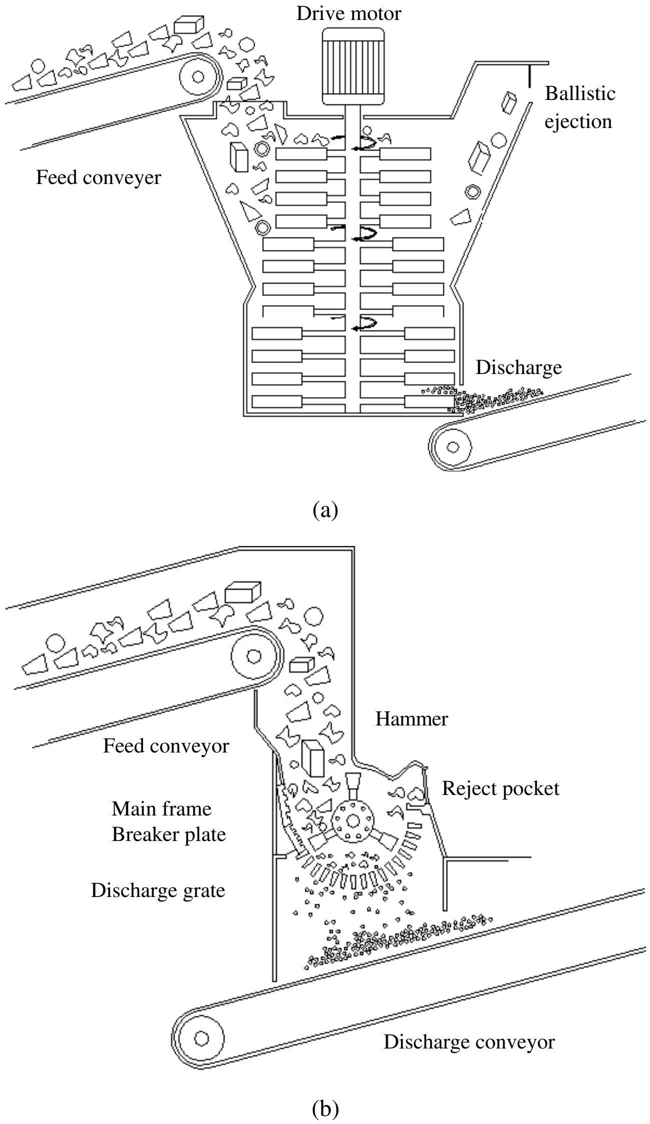


Figure 1. (a) Vertical Hammermill and (b) Horizontal Hammermill (Adapted from Vesilind and Rimer).[11]

or recyclable materials are progressively ground as they move downward through the machine.

Shear shredders are characterized by rotating blades or scissors that shear or cut material. Materials are fed to the shaft and they pass down through openings that form between the tops of opposing cutters and the opposite shafts. The size of the materials is reduced by shearing or cutting from the rotary blades or scissors. These units operate at a comparatively low rate of speed. Different size distributions of materials are always achieved after shear shredding.

A pulverizer, or glass crusher, uses a breaker plate and hammers to reduce the size of the materials. It is much like a hammermill, but without grates. This equipment has impact bars and impact plates that assist in the pulverization of glass and other materials. As these materials enter into the mill, they are hit by the hammers and thrown against the blocks where they are crushed into smaller pieces.

3.1.4. *Separation*

Separation is the operation in which materials are separated by their characteristics, which are size, shape, color, density, magnetism, and electrical conductivity. Due to the heterogeneity of typical solid waste and the diversity of recovery operations, separation is an important step in the recovery and utilization of solid waste. Several methods have been developed and some combination of methods is usually required to process a mixed waste stream. A list of these methods is provided in Table 3.

The separation can be divided into two systems; binary and polynary.[12] Binary is the separation system that acquires two output streams such as the waste separation of ferrous and nonferrous materials using a magnetic separator. If more than two output streams are obtained, the system is called polynary. For example, size separation using a screen with a series of different size holes is a polynary device.

Table 3. Separation Unit for the Recovery Processing.

Unit	Function	Preprocessing Requirement
Screening	Separation of over- and undersized material	Removal of bulky materials and large pieces of cardboard
Air classifiers	Separation of light combustible materials from waste stream by air	Removal of bulky materials, large pieces of cardboard, and shredding of waste
Magnetic separation	Separation of ferrous (magnetic) metals from nonmagnetic materials	Removal of bulky materials, large pieces of cardboard, and shredding of waste
Eddy current	Separation of nonferrous conductors (aluminum) from nonconductors (wood and plastic)	Removal of ferrous materials, bulky items, and shredding of waste
Float and sink separation	Separation of glass and aluminum	Removal of bulky materials

Source: Tchabanolgous Kreith.[17]

3.1.4.1. Binary separation system[12]

The efficiency of separation devices can be determined in terms of percentage recovery, purity, and effectiveness. In the binary separation system, the waste stream is composed of two components X and Y. The input to the binary separator is X_0 and Y_0. The output of exits in the first stream is denoted as X_1 and Y_1, and the second output stream is X_2 and Y_2. The sketch of this binary system is shown in Fig. 2

Assume that the device was intended to get the X in the first output stream and Y in the second stream. In practice, there might be some Y contaminated with X in the first stream. In the same way, the second stream is may be contaminated with some X. Thus, the calculation of the recovery percentage for the two components can be represented as:

$$\text{Recovery of component } X = \mathrm{R}(X_1) = \left(\frac{X_1}{X_0}\right) \times 100\% \tag{1}$$

$$\text{Recovery of component } Y = \mathrm{R}(Y_2) = \left(\frac{Y_2}{Y_0}\right) \times 100\% \tag{2}$$

The second term of performance evaluation is purity. It is needed in case the recovery of X is 100%, but no separation occurs because the device is not performing its desired function. The purity function represents the quality of material separated in terms of its contamination by the other material. This term is expressed as:

$$\text{Purity for component } X = \mathrm{P}(X_1) = \left[\frac{X_1}{X_1 + Y_1}\right] \times 100\% \tag{3}$$

$$\text{Purity for component } Y = \mathrm{P}(Y_2) = \left[\frac{Y_2}{X_2 + Y_2}\right] \times 100\% \tag{4}$$

Both a high-recovery and a high-purity percentages are the requirement for an ideal separator. The effectiveness of the separator was defined by Rietema[13] as:

$$\begin{aligned}\text{Effectiveness of component } X \text{ and } Y = \mathrm{E}(X, Y) &= \left|\frac{X_1}{X_0} - \frac{Y_1}{Y_0}\right| \times 100\% \\ &= \left|\frac{X_2}{X_0} - \frac{Y_2}{Y_0}\right| \times 100\% \end{aligned} \tag{5}$$

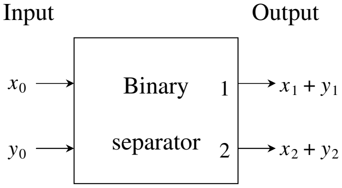


Figure 2. Schematic of Binary System for Waste Separation (Adapted from Vesilind and Rimer).[11]

This definition of effectiveness is rational in that if ideal separation occurs (all of X_0 goes to the first stream, so that $X_1 = X_0$ and $Y_2 = Y_0$), the effectiveness is 100%. Likewise, if no separation occurs ($X_0 = X_1$ and $Y_0 = Y_1$), then both measures of effectiveness are zero. It is noted that all X terms are in units of mass/time.

3.1.4.2. Polynary separation system[12]

There are two types of polynary separation systems. In the first system, as shown in Fig. 3, the waste stream is composed of X_0 and Y_0 and more than two streams are given as the output with X and Y appearing in all of them.

The recovery percentage can be represented as:

$$\text{Recovery of component } X = \text{R}(X_1) = \left(\frac{X_1}{X_0}\right) \times 100\% \tag{6}$$

The purity of the first stream can be expressed as:

$$\text{Purity for component } X = \text{P}(X_1) = \left[\frac{X_1}{X_1 + Y_1}\right] \times 100\% \tag{7}$$

The recovery of X in the mth output stream is

$$\text{Recovery of component } X \text{ in } m\text{th stream} = \text{R}(X_\text{m}) = \left(\frac{X_\text{m}}{X_0}\right) \times 100\% \tag{8}$$

The second type of polynary separation, the most common case is where the waste stream contains n components ($X_{10} + X_{20} + X_{30} \ldots X_{\text{no}}$) and these are to be separated in to m outputs. The schematic of this polynary separation is shown in Fig. 4.

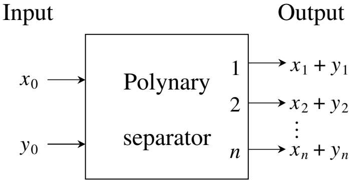


Figure 3. Schematic of Polynary System with Two Components in Waste Stream (Adapted from Vesilind and Rimer).[11]

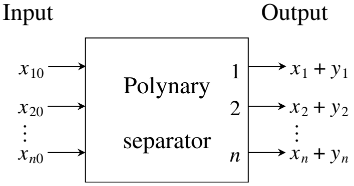


Figure 4. Schematic of Polynary System with n Components in Waste Stream (Adapted from Vesilind and Rimer).[11]

The notation X_{11} is the material that ended up in the first output stream, X_{12} is the second stream output, and so on.

$$\text{The recovery percentage of } X_{11} = \mathrm{R}(X_{11}) = \left(\frac{X_{11}}{X_{10}}\right) \times 100\%, \tag{9}$$

where X_{10} is the material X in the feed stream. The purity of the first stream in terms of X_1 is

$$\text{Purity for component } X = \mathrm{P}(X_1) = \left[\frac{X_{11}}{X_{11} + X_{21} + \cdots + X_{n1}}\right] \times 100\%. \tag{10}$$

Many types of separation are utilized as recycling techniques, which are as follow:

- Screening for size separation;
- Air classifier for density separation;
- Magnetic separator for ferrous and nonferrous materials separation; and
- Eddy current separator for conductive and nonconductive materials separation.

Each type of these techniques will be described in the following parts.

3.1.4.3. Screening

Screening is one type of material separation using the difference in particle size to characterize items. A screen of uniform-sized openings allows smaller particles to pass while rejecting larger sizes. Two material streams are obtained from these units that are extracts and rejects. Extracts are the materials that pass the opening with the desired size in underflow. Rejects are the materials that do not pass through the designed opening. The screen is the most important unit in solid waste processing, as it not only separates the waste stream, but also removes loose dirt and residue. The quality of recovered materials is consequently improved.

Different types of screens used in material processing include the vibrating screen, trommel or rotary drum screen, and the disk screen. Vibrating screens are used for free-flowing granular materials that are segregated fine particles. Trommel screens are used for large particle that easy to tie up the screen or attach with other materials. Disk screens are appropriate for rigid materials such as wood chips. The trommel screen is considerable the most popular screen for material processing.

A trommel screen is an inclined cylindrical, rotating screen that sorts material by size. It is mounted on rollers, with holes in the side as shown in Fig. 5. The cylinder sits at a downward slope in the order of 5° and rolls at slow speeds, 10–15 rpm. As the waste stream is conveyed to the screen, it tumbles through the cylinder as it rotates. The oversized materials, such as large plastic bottles, move through the cylinder. The undersized materials, such as can, small plastic bottles, and other small items, fall through holes in the cylinder. Trommel screens can be equipped with spikes or knives fastened to the inside of the cylinder to break open plastic bags. As a protection for

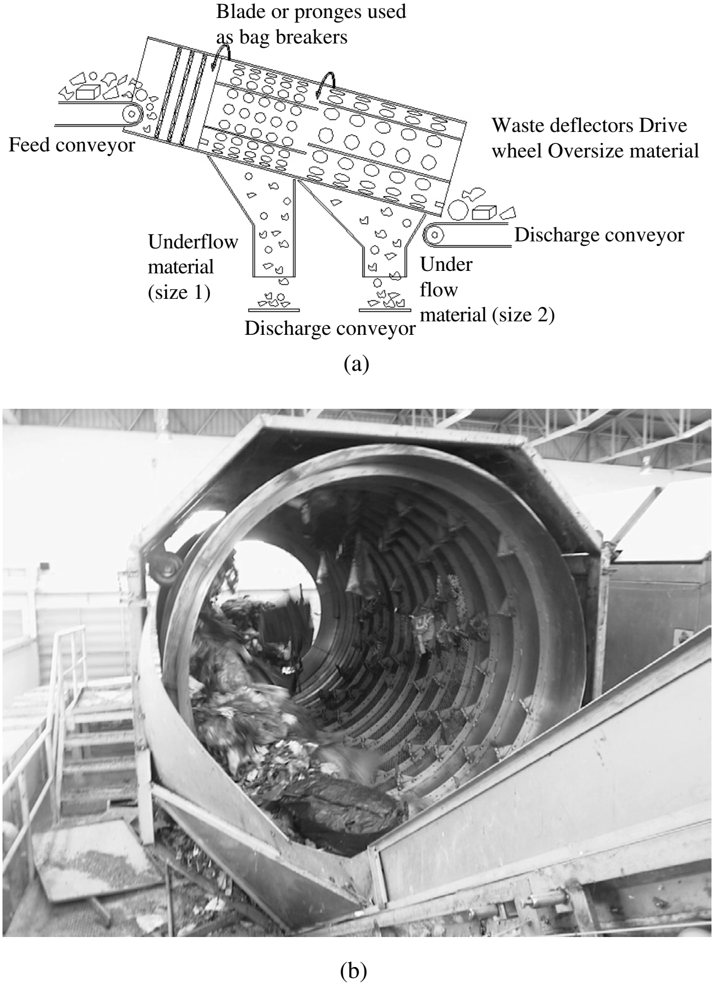


Figure 5. Trommel Screen for Waste Separation (a) Profile Diagram (11) and (b) Trommel Figure (Adapted from Taggart).[14]

glass breakage, the paddles, called lifters, are fastened to the inside of the cylinder to keep the material in the lower portion of the trommel.

The design specifications and operating parameters for trommel screens are dictated by the physical characteristics of the solid wastes or materials to be screened, the required feed rate and the separation requirement. The primary design parameters include trommel length, diameter, inclined slope, and hole size. The length of the trommel is a function of the material retention time. The complete removal of the required size from the solid waste stream can be achieved by using a trommel with a long length even though the majority of screening occurs in the first few feet of the unit.

The diameter of the trommel is another important parameter in design specification. A larger trommel diameter will result in a thinner bed of solid waste. One correlation developed for determining the diameter for given capacities is described by Taggart[14] as:

$$D = 7.66\sqrt{\frac{C}{G_\mathrm{s}}}, \tag{11}$$

where $D =$ trommel diameter (in.), $C =$ capacity (ton/hr), and $G_\mathrm{s} =$ specific gravity of the feed material.

Specific gravity values of various materials are provided in Table 4. It is important to note that this equation was developed for the screening of coal, sand, and stone-type materials and may not be appropriate for municipality solid waste, which is not as free-flowing a feed material.

The trommel rotation speed is also one of the important parameters for both the capacity and efficiency of the trommel unit. As the rotating speed increases, trommel efficiency passes through a maximum and then decreases sharply. The height of the trommel cylinder, which the wastes pass along before tumbling down, is determined by the critical speed for the particular unit. The critical speed is defined in Eq. (12):

$$v_\mathrm{c} = \sqrt{\frac{g}{4\pi^2 r}}, \tag{12}$$

where $v_\mathrm{c} =$ critical speed (rotation/s), $r =$ radius of the trommel cylinder (m), and $g =$ gravitational acceleration $= 9.81\ \mathrm{m/s^2}$.

Table 4. Specific Gravity of Various Materials in Solid Waste.

Material	Specific Gravity
Light materials	
Plastics	
— HDPE	0.96
— Polypropylene	0.90
— Polystyrene	1.05
— PVC	1.25
Paper	0.70–1.15
Cardboard	0.69
Wood	0.60
Heavy materials	
Aluminum	2.70
Steel	7.70
Glass	2.50

Source: Vesilind *et al.*[12]

The rotating speed normally ranges from 35% to 40% of the critical speed. The walls of the trommel can be made from wire mesh or can be punched-metal screens. The opening in the woven-wire screen can be either square or rectangular. Opening size, spacing, and shape will be dictated by the fraction identified for separation and the required feed rate. The screening efficiency of the trommel screen varies from 70% to 90%. Maintenance requirements for trommel screens normally include lubrication of bearings and removal of wire, ribbon, and cloth, which periodically wrap around the cylinder. In commingled waste processing, maintenance is required on a daily basis. However, this type of screen requires less cleaning than any other type of screen.

A vibrating screen is an inclined flat deck. The screen cloth, a wire mesh or solid metal plate, is powered by an electric motor to vibrate materials, which are free-flowing granular particles, to pass through the opening. The screen is often supported on springs and a motor turns an eccentric weight, which conveys movement to the material on the screen deck. Vibrating screens are suitable for screening fine particles. However, they are not appropriate for separation of large materials such as paper, which tend to bind over the screen openings.

Disk screens consist of sets of parallel, interlocking, rotating disks. The tops of the rotating disks form a moving surface that tumbles and conveys the solid waste. The materials from the waste stream fall between the disk spaces and oversized material (reject) are carried over the top of the disks. Changing the size and spacing of the disks can vary the opening size on disk screens. Since, the separation of material by disk screens depends on material size only, light and dense materials with the same size can be sorted out together. Air classifiers are generally needed after disk screens. Maintenance for disk screens includes the periodically unwrapping of cloth and stringy materials from around the disk. The periodically replacement of the disks that wear down is also necessary.

3.1.4.4. Air classification

Air classification is a technique widely used to separate materials based on their density and aerodynamic properties. Two components, light and heavy fractions, are divided using the density and aerodynamic criteria. The light fraction is composed of paper, plastics, and organics, and the heavy fraction includes metals, wood, and other dense inorganic materials. The materials to be separated are introduced into an air stream. Some materials are dragged in the air stream and taken away while others drop out. Whether a particle is carried by the flowing air stream to the light fraction or falls to the heavy fraction is determined by the forces exerted upon it. The two dominant forces acting upon the particle are the gravitational force and the drag force.

The drag force on a given particle may be expressed as:

$$D = {}^{1}\!/_{2}\,\rho_{\mathrm{a}} C_{\mathrm{D}} v_{\mathrm{a}}^{2} A, \tag{13}$$

where D = drag force, ρ_a = density of the air, C_{D} = drag coefficient for the material, v_{a} = air stream velocity, and A = surface area of the material exposed to the air stream.

While the gravitational force is expressed as:

$$W = v_s\,(\rho_{\mathrm{s}} - \rho_{\mathrm{a}})\,g, \tag{14}$$

where W = weight of the particle, v_{s} = velocity of the particle, ρ_{a} = density of the air, and g = gravitational force.

To suspend a particle in an air stream, the drag force must equal the gravitational force that may be expressed as (15):

$$v_{\mathrm{s}}(\rho_{\mathrm{s}} - \rho_{\mathrm{a}})g = {}^{1}\!/_{2}\,\rho_a C_{\mathrm{D}} v_{\mathrm{a}}^{2} A. \tag{15}$$

The velocity at which the drag force is equal to the gravitational force is referred to as the "terminal velocity." Solving the terminal velocity (V_{t}) as:

$$v_{\mathrm{t}} = \left(\frac{2v_{\mathrm{s}}(\rho_{\mathrm{s}} - \rho_{\mathrm{a}})g}{\rho_{\mathrm{a}} C_{\mathrm{D}} A} \right)^{0.5} \tag{16}$$

If the drag force exerted on any particular particle in the air stream is greater than the gravitational force on that particle, the particle will be lifted and carried along with the air stream. On contrary, if the gravitational force is greater than the drag force, the particle will fall from the air stream.

For Eq. (17), if the average dimension of the particle is applied instead of surface area, and the particle is a perfect sphere, then:

$$A = \frac{\pi d^2}{4} \quad \text{and} \quad V = \frac{\pi d^3}{6}, \tag{17}$$

where V = volume of the particle (m^3), A = projected area of the particle (m^2), and d = diameter of the particle (m).

To suspend a particle in an air stream, the drag force must equal the gravitational force where the acceleration is gravitational ($a = g$), then the equation of this event may be expressed as:

$$v = \left[\frac{4(\rho_{\mathrm{s}} - \rho_{\mathrm{a}})gd}{3\mathrm{C}_D \rho_{\mathrm{a}}} \right]^{1/2}, \tag{18}$$

which is called Newton's law. Assuming that laminar flow conditions occurs during operation, and substitution the Reynolds number into Eq. (19), Stokes law is obtained as:

$$v = \frac{d^2 \mathrm{g}(\rho_{\mathrm{s}} - \rho_{\mathrm{a}})}{18\mu}, \tag{19}$$

where v = velocity (m/s), ρ_{s} = particle density (kg/m^3), ρ_{a} = density of the air (kg/m^3), and μ = viscosity (kg/s-m).

As a precaution for use of Eq. (20) for an air classified, it must be noted that, in practice, a differing turbulence flow is expected, as in this equation where laminar flow is assumed.

Four participle shapes were utilized in the air classification model: flakes, cylinders, splinters, and irregular-shaped particles (which are generally modeled as cubes). The characteristics of various particle shapes are shown in Table 5.[16] Material properties and typical terminal velocities calculated for various waste components are presented in Table 6. Efficiency of an air classifier can be described by

Table 5. Characteristics of Particle Shapes Utilized in Air Classification.

Particle Designation	Aerodynamic Model	Dimensional Characteristics	Drag Coefficient (C_D)	Cross-Sectional Area (A)
Flake	Flat plate	$t << L$	1.0	L^2
Cylinder	Cylinder	$L < 15D$	0.7	LD
Splinter	Cylinder	$L > 15D$	0.9	LD
Irregular	Cube	All sides of equal L	0.8	$2^{0.5}L^2$

Source: Savage *et al.*,[16]
Notes: $L =$ length, $t =$ thickness, and $D =$ diameter.

Table 6. Material Properties and Typical Terminal Velocities for Various Waste Components.

Waste Component	Particle Designation	Particle Density (kg/m^3)	Typical Terminal Velocity (m/s)
Newsprint	Flake	560	1.0
Polyethylene film	Flake	912	4.4
Polyethylene	Irregular	912	8.7–15.3
PVC film	Flake	1,008	0.6
Lumber	Splinter	480	2.2–8.5
Plywood	Flake	552	5.9
Textiles	Flake	242	2.3
Rubber	Irregular	1,773	18.0
	Flake	1,773	8.4–12.0
Aluminum	Flake	2,688	2.4–4.6
	Irregular	2,688	9.8–44.2
Aluminum can	Cylinder	58	6.6
Ferrous	Flake	7,840	4.0–5.9
	Irregular	7,840	16.6–75.0
Ferrous can	Cylinder	144	9.9
Glass	Irregular	2,400	2.9–22.5

Source: Savage *et al.*,[16]

the following expression[17]:

$$\text{Efficiency} = \sqrt{\frac{X_e}{X_0} \times \frac{Y_r}{Y_0}} \times 100\%, \tag{20}$$

where X_e = mass of combustible light materials extracted from the air classifier, X_0 = mass of combustible light materials input to the air classifier, Y_r = mass of noncombustible materials exiting the air classifier as heavies, and Y_0 = mass of noncombustible materials input to the air classifier as heavies.

Another method to determine the effectiveness of air classification is by measuring the aerodynamic velocities of the particles to be separated. This determination method is called the drop test.[12] The representative particles to be separated are dropped from a reasonable height in the absence of air turbulence, and the time needed to reach the ground is measured. The results of this test for various materials, paper, plastic, aluminum, and steel, are presented in Fig. 6. With the increasing of air velocity, different amounts of extracted material are obtained as shown in Table 7. The drop test is a useful method for obtaining information for an each material to be extracted.

Air classifiers are categorized into three major types, which are vertical column, horizontal, and air knife classifiers.

In vertical air classifiers, the material is fed from the top and air is introduced from the bottom of the classifier. A variety of designs has been utilized to increase the turbulence and residence time of the column. With high turbulence and longer

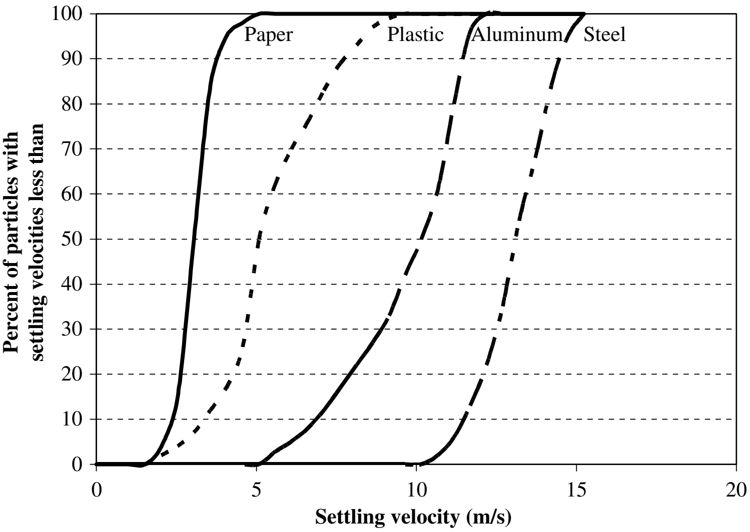


Figure 6. Results of Drop Test for Air Classification of Various Materials (Adapted from Vesilind *et al.*).[12]

Table 7. Percentage of Material Extracted from the Drop Test.

Air Velocity (ft/min)	Percentage of Extracted Paper	Percentage of Extracted Plastic	Percentage of Extracted Aluminum	Percentage of Extracted Steel
700	80	10	0	0
1,000	100	50	0	0
2,000	100	100	50	0
3,000	100	100	100	95
4,000	100	100	100	100

Source: Vesilind *et al.*[12]

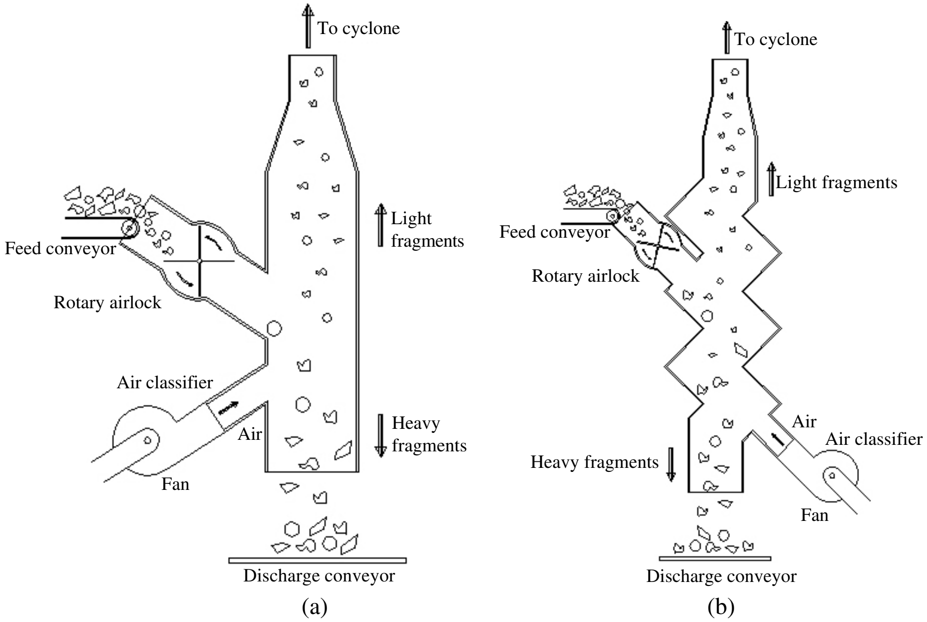


Figure 7. (a) Vertical Air Classifier and (b) Zigzag Classifier (Adapted from Tchabanolgous and Kreith).[17]

residence time, the quality of material separation is improved. The vertical air classifier is shown in Fig. 7(a). Additional designs include zigzags, baffles, and other obstacles. The zigzag is the most popular modification to vertical air classifiers. In a zigzag classifier, the vertical flow column consists of a series of 90° or 60° turns as shown in Fig. 7(b). This design was developed to allow multiple stages of classification to improve the quality of separation. Material tumbles as it drops through the air current falling from one shelf onto the next. With the turbulent vortex, bunches of material are broken up and the light particles are transferred to the upward air stream. The heavy particles drop from vortex to vortex until they are removed at the bottom. The dropping action also helps separate any agglomerated particles. A baffled column was developed to allow multiple stages of classification to occur, improving the quality of separation. This type of air classifier is similar to a vertical air classifier, except that materials tumble down deflectors as they drop through the air current, falling from one shelf onto the next.

In a horizontal air classifier, material is fed by conveyor and dropped into a horizontal air stream. Heavy objects that are more affected by gravitational force than by the pneumatic forces of the air current, drop through the air current. The lighter fraction is carried farther distances by the air stream or is carried away with the air stream into a cyclone that acts to separate the light entrained matter from the air stream itself. Materials that are not dropped out of the airstreams by the cyclone mechanical separation are later separated from the air stream using either a bag or a wet scrubber.

The air knife classifier is similar in concept to a horizontal air classifier. The small difference is that the air knife introduces a sharp curtain of air through which the material passes. The air is injected horizontally through a vertically dropping feed as shown in Fig. 8. The light materials are carried with the air stream while the heavy ones, with sufficient inertia to resist a change in direction, drop through the air stream. Air knives are designed for a fine separation of materials of different densities. Maintenance for air knives consists of separating the heavy fraction such as forgings, castings, and rolled stock from the light fraction such as light-gauge metals.

3.1.4.5. Magnetic separation

Magnetic separation is the operation in which ferrous (magnetic) materials are separated from nonmagnetic material through their magnetism property. Magnets can be

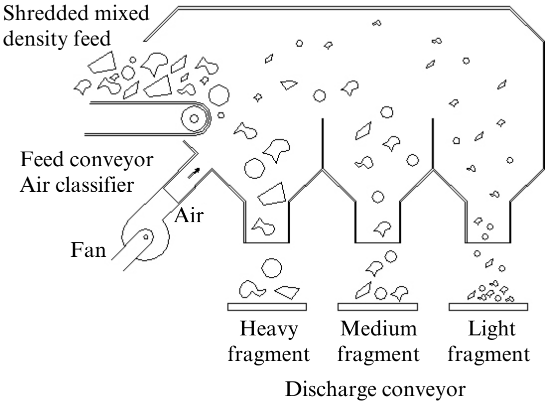


Figure 8. Horizontal Air Knife (Adapted from Tchabanolgous and Kreith).[17]

categorized into two types: (1) electromagnets, which use electricity to magnetize or polarize an iron core, and (2) permanent magnets, which utilize permanently magnetized materials to create a magnetic field. The general process of magnetic separation uses a belt magnet installed above a conveyor belt, with the belt magnet moving across the conveyor transporting materials. Ferrous materials such as steel cans are extracted from the waste stream and stick onto the base of the belt covering the electromagnet. The belt moves these materials off to the side where they cease to be under the power of the magnetic field and drop off the conveyor. Efficient magnetic separation is affected mostly by the extent to which ferrous materials are physically liberated from other types of materials and the degree of size reduction provided. In any event, the primary requirement to achieve high efficiency in magnetic separation is that the materials be physically freed from one another, resulting in minimum contamination of recyclable products. Size reduction through hammermills, shredders, and crushing is considered to be satisfactory preparation for efficient magnetic separation of ferrous materials from the solid waste stream. A strict particle size in the feed material to the magnetic separator is not crucial due to the fact that the magnetic equipment can accommodate nearly all ferrous objects ranging from large tin cans down to small metal scraps. However, occasional large items such as appliances need to be reduced in size to pieces.

The three major types of magnetic separation are the suspended belt magnet, the head pulley, and the suspended drum.

The suspended belt magnet is a fixed magnet located between the pulleys of a conveyor (Fig. 9). The magnetic conveyor is suspended above the flow of the waste stream, either parallel or transverse to the flow. Typically, as the flux density and flux gradient at various locations across the feed conveyor vary; the magnetic strength varies accordingly. The result of these magnetism properties is that the ability to lift ferrous materials off the belt in any location of the conveyor varies. During the separation process, magnetic metals are often sheltered by a burden of nonmagnetic material. This overburden can also be lifted up by the strength of force exerted

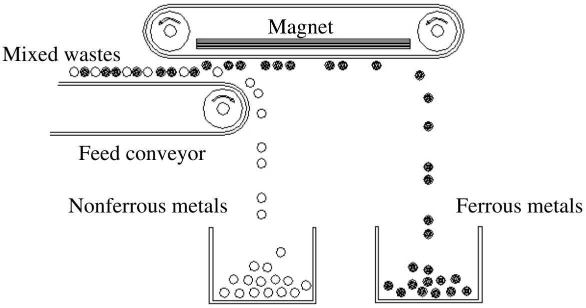


Figure 9. Parallel Suspended Belt Magnet (Adapted from Tchabanolgous and Kreith).[17]

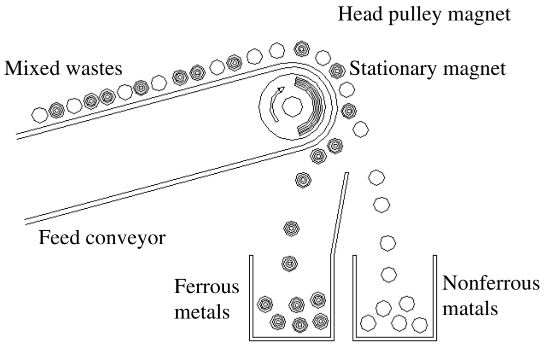


Figure 10. Head Pulley Magnet (Adapted from Tchabanolgous and Kreith).[17]

between the magnetic material and the magnet itself. The nonmagnetic materials can thus be lifted and conveyed to the magnetic belt, resulting in contamination of the recyclable products.

The head pulley magnet is installed at the forward pulley in the conveyor. It is an integral part of a belt conveyor as seen in Fig. 10. As the waste stream reaches the forward pulley, magnetic materials fall through the free-fall zone and are dropped on one side of the splitter, while nonmagnetic materials, unaffected by the magnet, travel on the other side of the splitter. Without the effects of lift or tumble, the under-burden material, such as plastic, paper, and wood chip, which is entrapped under the magnetic metal is carried over into the magnetic product resulting in the contamination of the recyclables.

The suspended drum magnet consists of a stationary magnet inside a rotating cylinder. Magnetic materials are pulled to the front of the drum, held against it, as it travels and then discharged free of the nonmagnetic materials. In this system, the separation occurs while the feed stream is suspended in air and operated upon simultaneously by momentum, gravity forces, and magnetic forces. The advantage of this suspended drum magnet resulting from the air-separating activity is that the nonmagnetic materials are less entrapped in the magnetic materials stream than in other magnetic separation system. A schematic diagram of this magnet installation is shown in Fig. 11.

3.1.4.6. Eddy current separation

Eddy current separation is an effective way of removing nonferrous metals from streams of industrial or municipal waste.[18] The process is widely used to separate aluminum and copper and to remove metals from recycled glass. The separation technique is based on the fact that conductive materials resist being moved in a gradient magnetic field and, vice versa, will accelerate in a moving gradient field.

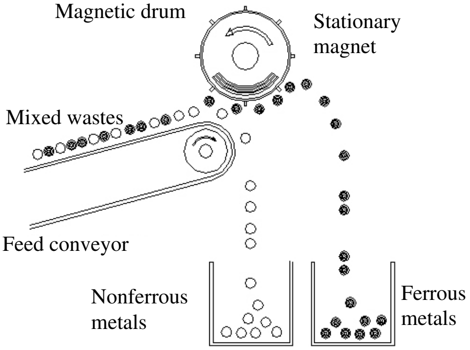


Figure 11. Suspended Drum Magnet (Adapted from Tchabanolgous and Kreith.)[17]

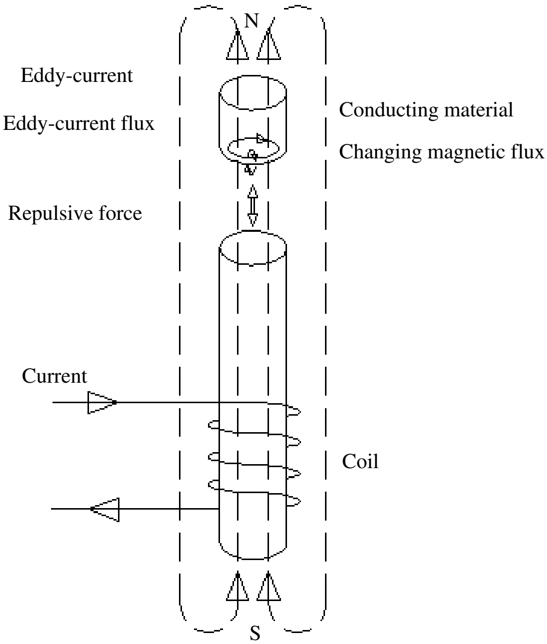


Figure 12. Principles of Eddy Current Phenomena (Adapted from Rem *et al.*).[18]

The principles of the eddy current phenomena are shown in Fig. 12.[12] When an electric current is passed through a coil surrounding a core made of a conducting material, a magnetic flux develops. By varying the current, one may cause the magnetic flux to change. This changing magnetic flux is the key to the eddy-current phenomena. When a magnetic field in a conducting medium changes with time, an electromotive force is generated in the plane of the medium, which is perpendicular to the direction in which the flux is changing. As a result of the electromotive

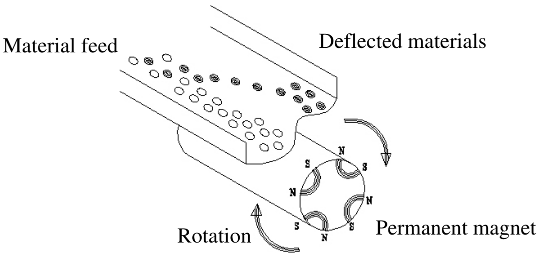


Figure 13. Eddy Current Separator (Adapted from Rem *et al.*).[18]

force, eddy currents develop within the material. This eddy-current principle can be employed to separate various nonmagnetic conductive materials from nonconductive materials. In general, all conductor materials resist changes in magnetic field strength. The repulsive forces between the primary and secondary magnetic fields are utilized in eddy-current separators to make a separation between materials. The repulsive force exerted on a particle increases with the fourth power of the material radius, and thus bigger particles are much more affected than small particles. The principal criteria used to make the separation are related to conductivity and mass. Shape and thickness can also be factors in eddy current separation. Eddy current separators create forces on conductors but not on nonconductors. Based on this principal, nonmagnetic metals may be separated from plastics, wood, rubber, etc.

Figure 13 shows a schematic of one type of eddy-current separation.[19] An eddy-current separator consists of a drum covered with magnets that are oriented alternately N–S and S–N. The fluctuating field of the spinning drum induces eddy currents in electrically conductive particles moving close to the drum. The conductive particles present in the field are then expelled along the direction of rotation. On the other hand, poorly conductive materials stay on the belt and drop close to the drum. The trajectory of particles is generally determined by a combination of gravity, the frictional forces of the transportation belt and the air and the electromotive force.

3.2. *Chemical Recycling Method*

Chemical recycling is widely used in plastic recycle process. Thus, this method is usually refer to advanced technology processes that convert plastic materials into smaller molecules such as liquids or gases, which can be used as a feedstock for petrochemical and plastic production.[20] In this process, the waste is changed to be a high product yield by depolymerization processes. Advanced chemical processes under the category of chemical recycling appear as pyrolysis, gasification, liquid–gas hydrogenation, viscosity breaking, steam or catalytic cracking, and the use of plastic solid waste as a reducing agent in blast furnaces.

Several chemical recycling methods currently used are:

- direct chemical treatment by blast furnace,[21]
- direct treatment by coke oven,[22]
- liquefaction,[23]
- polymer condensation,[24] and
- pyrolysis and gasification.[25]

The main advantage of chemical recycling is the possibility of treating heterogeneous and contaminated polymers with limited use of pretreatment. Among many chemical recycling methods, pyrolysis becomes a most widely used technique.

The pyrolysis process is the chemical decomposition of condensed wastes by heating that occurs spontaneously at high temperatures. The hydrocarbon content of the waste is converted into a gas, which is suitable for utilization in either gas engines, with associated electricity generation, or in boiler applications without the need for flue gas treatment. It has the ability to produce a clean and high calorific value gas from a wide variety of waste and biomass streams. This process is capable of treating many different solid hydrocarbon based wastes whilst producing a clean fuel gas with a high calorific value. This gas will typically have a calorific value of 22–30 $\mathrm{MJ/m^3}$ depending on the waste material being processed.[20] The lower calorific value is associated with biomass waste and the higher calorific value being associated with other wastes such as sewage sludge. If waste contains significant quantities of synthetic materials such as rubber and plastics, higher calorific values can be obtained from the produced gases. Solid char is also produced from the process, which contains both carbon and the mineral content of the original feed material. Recently, pyrolysis has been investigated as a promising method for solid waste recycling especially for plastics,[26] biomass,[26,27] and rubbers.[28,29]

4. Materials Recovery Facilities

Characteristically, several facilities are involved in recycling processes. Some facilities operate as receiving and shipping centers such as drop-off facilities, which include no processing separation equipment. If a drop off pays collectors for recyclables, it is called buy back center. If the facility contains processing equipment, it is called an intermediate processing center. The major difference between each type of facility is the separation equipment. For example, drop-off and buy back centers deal directly with the public and they simply accept various materials and ship those materials to more complex processing facilities. They do not typically contain significant processing equipment. In addition, if materials are source-separated, the facility does not need sorting equipment. In contrast, if a facility receives commingled solid waste or recyclables, separation equipment is needed as well as equipment for preparing the materials for market. Such a facility is commonly called a “material recovery facility (MRF).” In general, MRFs are buildings with equipment to sort recyclable materials and convert them into a saleable form. The equipment provided in MRFs depends on the purpose of each MRF.

The objectives of a commingled MRF are to receive, process, and ship materials to market or end users. An MRF can be used to separate and process source-separated materials, as well as separate materials from commingled solid waste. Materials from source separation are generally mixed together. These materials typically consist of glass, high-density polyethylene (HDPE) and PET plastics, aluminum, and ferrous and nonferrous metals. Generally, for the materials received from source separation, old newspaper is kept separate from other materials. The distinctiveness of source-separated materials to be processed generally dictates two separate process lines. The first line is designated to process the light fraction recyclables that are newspaper, cardboard, mixed paper, and plastic film. The second line is equipped with machinery to separate the heavy fraction materials, which are glass, aluminum, ferrous and nonferrous metals, wood, rubber, and plastics, from the waste stream.

In Fig. 14, a typical separation process design is shown for a commingled MRF utilizing both manual and mechanical separations of materials.[30] Commingled solid waste from residential and other sources is discharged into the receiving area. The first stage of waste operation is presorting to remove recyclables, reusable, and oversized material; for example, cardboard, white goods, broken furniture, and lumber. After presorting the commingled waste is loaded onto the conveyor to transfer the waste stream to each separation unit. Manual sorting is the first operation unit to separate additional large items from the see-through plastic bags on the conveyor as the second-stage presorting station. The bag-breaking is the next step after hand sorting. This operation can be done either manually or mechanically. In some operations, the hand sorting to separate plastics or other specific materials is arranged correspondingly after the bag-breaking machine. Material remaining on the conveyor is moved into a trommel screen or disc screen for size separation. The over- and undersized materials are separated. The undersized materials from trommel screen are either hauled away for disposal in a landfill or processed further for combustion or used to produce compost. The oversized or remaining materials are transferred for further processing by magnetic separation used to separate ferrous and nonferrous materials, and followed by the shedding operation. The recyclables might be sent to balers to reduce volume for marketing.

Other features may be needed for MRF operations include containers for recyclable materials, platform scale, glass storage bunker, temporary storage containers, and mixed-waste receiving hopper.

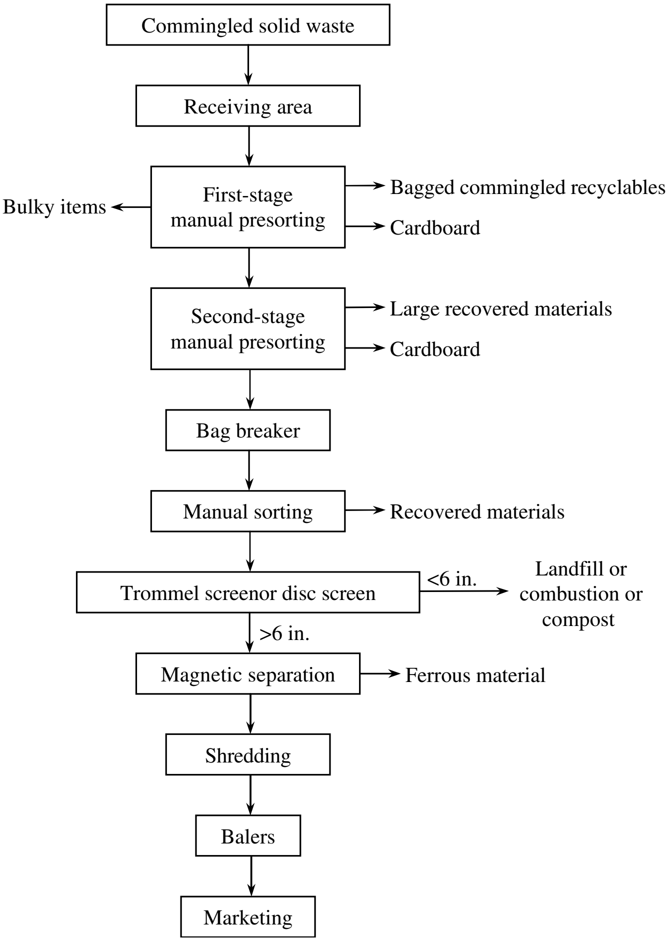


Figure 14. Layout Example of Commingled MRF (Adapted from Tchabanolgous and Kreith).[17]

5. Marketing of Recyclable Materials

Merely collecting and processing recyclables are not considered recycling — these recovered materials must be reutilized as the source materials for new modified products. Thus, feasible markets for the sale of recyclable materials, which are retrieved in curbside collection and drop-off and buy-back centers, are essential for the success of a community's recycling program. The key to obtaining markets for recyclables is the quality of recovered materials. It is necessary to segregate the materials into various grades with minimal contamination. The materials must have consistency in quality. Moreover, to make sustainable marketing of recyclable materials possible, it is necessary to keep the supply and demand of materials in balance.

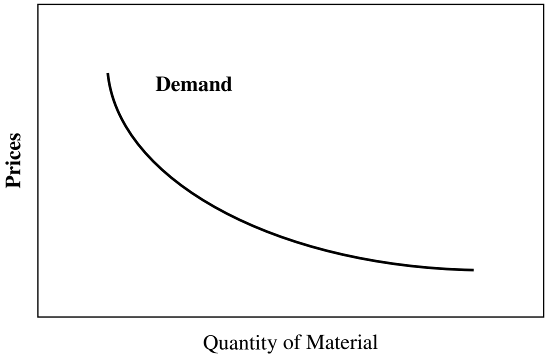


Figure 15. Demand Curve for Recyclable Materials Utilized by Industry to Produce Final Products (Adapted from Rogoff and Williams).[30]

Demand of recyclable materials to produce new products is dependent largely upon production capacity, manufacturing capital costs, contamination problems, and other costs related with using reclaimed material. The demand curve for recyclable materials utilized by the industry to produce final products is shown in Fig. 15.[30] This graph demonstrates how much of a particular recyclable material the industry is willing to purchase at a provided price. As the quantity of materials increases, the purchase prices obviously decrease.

To accept recyclable materials for industrial processes, many manufacturers have complete manufacturing processes for the raw materials and may be unwilling to change these processes for the acquired recyclables. High capital costs are required to expand production capacity for these recyclables. Other costs associated with the use of recycled material are the increased inspection costs, which are accumulated during every step of the manufacturing process from the beginning to the approval of the final product. As a result of these additional costs, many of the products produced from recyclables are more expensive than those produced from virgin materials. The contamination of recyclable materials to some degree limits the recycling of materials. Although raw materials can be both virgin and recycled materials, the purchaser may be wary of potential contamination problems. The buyer may set stringent standards before purchasing recycled material.

The supply curve of recyclable materials delivered to manufacturers is shown in Fig. 16.[30] This graph illustrates the quantity of recyclable material that suppliers are willing to provide at every possible price during a given period. The supply of recycled material is shown to be very flexible due to relatively easy and inexpensive gathering and selling of materials. As a result, the material prices are changed over short periods. Moreover, the amount of obtainable materials varies seasonally. For example, more beverages are consumed in summer months than in winter months resulting in higher amounts of glass and aluminum beverage containers during the

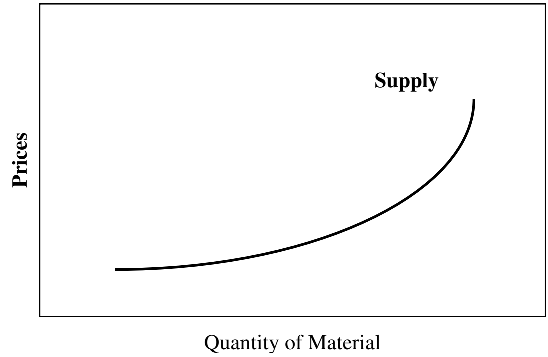


Figure 16. Supply Curve of Recyclable Materials Delivered to Manufacturers (Adapted from Rogoff and Williams).[30]

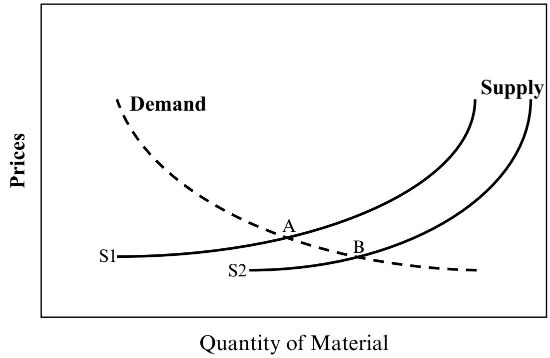


Figure 17. Change in Demand and Supply Curve as Recycling Program is Performed (Adapted from Rogoff and Williams).[30]

summer months. In that case, the availability of certain recycled materials fluctuates. This fluctuation may make recyclable materials less attractive to an industry that is looking for a stable, consistent supply of material.

Nowadays, many communities try to introduce recycling programs. Recycling and public awareness programs to promote recycling and reuse have been set up. The results of these programs directly effect a dramatic increasing amount of recyclable materials available for industry. This phenomenon can be described graphically as shown in Fig. 17. Point “A” represents a typical equilibrium point at which demand and supply are in balance for an original supply. With the initiation of the recycling program, the original supply curve S_1, is shifted right to S_2. Consequently, the equilibrium point moves from “A” to “B.” The new equilibrium point B shows a slight increase in the quantity of reclaimed materials and a dramatic reduction in the price paid for those materials.

5.1. *Recycling Program Development*

A recycling program affects a major change in the volume of waste for solid waste management. Basically, it involves a change in handling waste for most residents. Residents must know what is expected of them and must help make the program a success. The conveniences of the recycling process must be made clear to the residents. Indeed, some studies show that the perception of inconvenience of recycling was stronger among surveyed respondents who did not recycle than among those who did.[10,31] To make participation as convenient as possible, and thus maximize the amount and the quality of material collected, communities[31,32]:

- provide curbside collection of recyclables;
- provide seasonal and frequent curbside collection of yard trimmings;
- offer service to all households;
- utilize collection methods that encourage resident participation as well as yield high-quality and readily marketable materials (such as using large clear plastic bags or bins for commingled food and beverage containers, and separate set-outs for paper grades);
- provide adequate containers for storage of residential recyclables; and
- establish recycling drop-off or buy-back centers.

The previous study reported that the success of solid waste management program depends on waste minimization behavior that is likely to be influenced by a concern for the environment and the community, and is likely to be inhibited by perceptions of inconvenience and lack of time and knowledge.[33]

To ensure success, a community recycling program must be well developed. First, communities should decide which materials should be recycled. This decision should be derived from an analysis of the volume of the community's recyclable materials that can be diverted to the recycling operation and the marketability and economics of material handling. Once it is known which material will be collected and in what volume, decisions can be made concerning how to collect the material, what processing will be needed, and how much processing and storage space will be required. The needs of potential buyers will help clarify what types of equipment for processing and storage will provide better marketability. A well-designed recycling operation should have minor environmental impact. Significant effort must be paid to operate the facility as a good neighbor and keep nuisance conditions from developing. Definitely, a recycling program must comply with the requirements of state recycling registration. A successful regional recycling program is possible through conscious solicitation of cooperation among states, municipalities, commercial businesses, schools, and citizen volunteer groups and organizations. Effort should be made to reach out and involve this groups.[34]

For demonstrating and draw resident participation, a pilot-recycling program is needed. In a pilot program, recyclables are collected using prescribed methods for a certain period. Then, the efficiency of this program is evaluated. A pilot program allows the community to try an approach, such as clear bag collection or bin collection, without the expense of going community wide. With an education program for the public, the pilot program can initiate public discussion and an understanding of the recycling program. A further advantage of this program is the opportunity to obtain an estimation of the expected recyclable quantity.

Besides the pilot program, the voluntary recycling program is another method to initiate community recycling. A voluntary program can be used to educate people concerning the requirements and benefits of recycling without the enforcement of a mandatory recycling ordinance. Once residents are familiar with the voluntary program and many are already participating, the voluntary program can be shifted to the mandatory program. Changes in procedures can also be made more easily when the program is voluntary than when enforcement is associated with noncompliance. For many communities, the high participation rates achieved with an efficient and well-publicized voluntary program have eliminated the need for a mandatory program. With the initiation of these recycling programs, the success of solid waste recycling is likely to be successful.

5.2. *Market Development for Recyclables*

To achieve the recycling goal, market development should be planned. In general, market development is a complex process that requires the careful incorporation of government, the public, and private industry across a strong set of policies and programs. Market development can only succeed, if consumers purchase the products that have a recycled content. Securing stable and reliable markets requires:

1. basing market decisions on a clear understanding of the recyclables market system and
2. sharing decision making among recycle program planners, government officials, the public, and the private sector.

It is frequently said that the ultimate success of recycling depends on stable and reliable markets for recyclables. Unless a community has markets for the materials it collects, it may end up temporarily storing some materials and later landfilling some or all of them. There are two processes available to secure a stable and reliable market for recyclables. First, it requires marketing decisions based on a clear understanding of the infrastructure of recycling are required. Second, recycling program planners, government officials, and the public are required to share responsibility with the private section in adopting and implementing market development strategies.[35]

A market is an institution that serves as a link between buyers and sellers of a particular good. In recycling, the market infrastructure includes two groups: intermediate markets and end-use markets.

Intermediate markets are commonly classified as collectors, processors, brokers, and converters.

- Collectors are companies that collect recyclables or are waste haulers who have expanded their business to include collecting recyclables from residents and businesses.
- Processors include local, private buy-back centers, and privately or publicly operated MRFs. Processors accept and modify recyclables from residential or business sources by sorting, baling, crushing, or granulating.
- Brokers buy and sell recyclable materials, often arranging to have them shipped from one location to another by collectors or processors. The broker receives a fee for this service. Some brokers provide processing services, while others only move preprocessed recyclables.
- Converters are companies that take recyclable materials in a raw form and change them; therefore, they are readily useable by a manufacturer. An example of a converter is a company that produces pulp from paper; the pulp is then used by a paper mill.

End-use markets are public-– or private-sector entities that purchase recovered materials from a number of sources and use those materials as feedstock to manufacture a new product. Communities may want to market some materials directly to end-use markets. Although direct marketing eliminates the need to pay a broker, the community assumes the risk if the buyer rejects a slightly contaminated load and there is no alternative market readily available.

The product consumer also plays a role that affects the market development. Without consumers willing to buy products with a recycled content, market development will fail. Usually, consumers can convince end-users to utilize a recycled content as in the raw materials for their preferred product. As a result, the end-user improves the quality of recycled-content products. Awareness of the public in buying products and packages with recycled content then becomes one of the major factors that effect the recyclable markets, especially industries receiving recyclables for their manufacturing. Nowadays, several large companies are promoting the recycled content of their plastic bottles, paperboard boxes, and other packages. The increase of consumer purchasing preference has become one method to develop recyclable markets. The promotion of recycled content is possible by creating a need for standards for product labeling. The subject of labeling issues is currently being addressed by many countries. Labeling policies should reward those manufacturers who make a commitment to using recycling content. The preferable labeling is a full disclosure type labeling that would fully describe the recycled content, indicating the specifics about the amounts and types of recyclables. In addition, labeling should be internationally consistent. However, the additional expenses coming from labeling processes can prohibit end-users from labeling products.

6. Design Examples and Questions

6.1. *Example 1*

A binary separator has a feed rate of 1.1 ton/h. The first and second output streams are 500 and 600 kg/h, respectively. It is measured that 450 kg of X appears in the first stream and 30 kg of X ends up in the second stream. Calculate the percentage recovery, purity, and effectiveness of this binary separator and sketch the schematic of this system.

Solution:

1. Percentage recovery of X in the first stream is:

$$\mathrm{R}(X_1) = \left(\frac{X_1}{X_0}\right) \times 100\% = \left(\frac{450}{450+30}\right) \times 100\%.$$

$\therefore$ The percentage recovery of X in the first stream $= 93.75\%$.

2. Purity of this output stream is:

$$\mathrm{P}(X_1) = \left[\frac{X_1}{X_1+Y_1}\right] \times 100\% = \left[\frac{450}{450+50}\right] \times 100\%$$

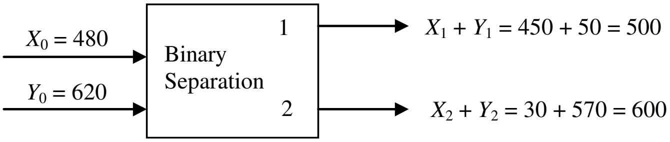
$\therefore$ The purity of output stream $= 90\%$.

3. Effectiveness of the separation

$$\mathrm{E}_{\mathrm{x,y}} = \left|\frac{X_1}{X_0} - \frac{Y_1}{Y_0}\right| \times 100\% = \left|\frac{450}{480} - \frac{50}{620}\right| \times 100\%.$$

$\therefore$ The effectiveness of the separation $= 85.69\%$.

4. Schematic of the binary separation



6.2. *Example 2*

An air classifier has a light fraction of 700 ton/day and a heavy fraction of 500 ton/day coming to the system. If the separator has percentage recoveries for both light and heavy fractions of 95%, calculate the following items:

1. extracted portion of light and heavy fractions,
2. purity of light and heavy fractions,

3. the efficiency of air classifier, and
4. sketch the schematic of this system.

Solution:

1. Find the extracted portion of the light fraction.

$$\text{Percentage recovery of light fraction } = \mathrm{R}(X_1) = \left(\frac{X_1}{X_0}\right) \times 100\% = 95\%$$

$$X_1 = \frac{95 \times X_0}{100} = \frac{95 \times 700}{100} = 665\,\text{ton/day}$$

The extracted light fraction from the waste stream is 665 ton/day.

2. Find the extracted portion of the heavy fraction.

$$\text{Percentage recovery of heavy fraction } = \mathrm{R}(Y_2) = \left(\frac{Y_2}{Y_0}\right) \times 100\% = 95\%$$

$$Y_2 = \frac{95 \times Y_2}{100} = \frac{95 \times 500}{100} = 475\,\text{ton/day}$$

The extracted heavy fraction from the waste stream is 475 ton/day.

3. Purity of the light fraction.

$$\text{Purity of light fraction } = \mathrm{P}(X_1) = \left[\frac{X_1}{X_1 + Y_1}\right] \times 100\%$$

$$\mathrm{P}(X_1) = \left[\frac{665}{665 + 25}\right] \times 100\% = 96.37\%$$

The purity of light fraction is 96.37%.

4. Purity of the heavy fraction

$$\text{Purity of heavy fraction } = \mathrm{P}(Y_2) = \left[\frac{Y_2}{X_2 + Y_2}\right] \times 100\%$$

$$\mathrm{P}(X_1) = \left[\frac{475}{475 + 35}\right] \times 100\% = 93.14\%$$

The purity of heavy fraction is 93.14%.

5. The efficiency of the air classifier

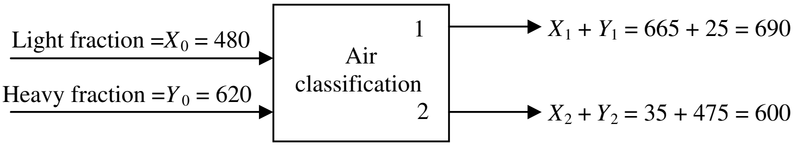
$$\begin{aligned}\text{Effectiveness of air classifier} = \mathrm{E}(X, Y) &= \left|\frac{X_1}{X_0} - \frac{Y_1}{Y_0}\right| \times 100\% \\ &= \left|\frac{X_2}{X_0} - \frac{Y_2}{Y_0}\right| \times 100\% \\ \mathrm{E}(X, Y) &= \left|\frac{665}{700} - \frac{25}{500}\right| \times 100\% = 90\%\end{aligned}$$

or

$$\mathrm{E}(X, Y) = \left|\frac{35}{700} - \frac{475}{500}\right| \times 100\% = 90\%$$

The effectiveness of air classifier is 90%.

6. Schematic of the air classification system



6.3. *Example 3*

Two materials are subjected to separation by air classification. By calculation, the first item has a drag force greater than the gravitational force, while the second one has a drag force less than the gravitational force. What will happen when these two items are conveyed into the air classifier?

Solution:

From theory, a particle can remain suspended in the air stream when the drag force is equal to the gravitational force. If the drag force exerted on any particular particle in the air stream is greater than the gravitational force on that particle, the particle will be lifted and carried along with the air stream. On the contrary, if the gravitational force is greater than the drag force, the particle will fall from the air stream. Thus, the first item will hang and be conveyed along with the air stream and the second item will fall from the air to the hopper.

6.4. *Example 4*

Trommel screening is utilized to separate plastics from the waste stream. The required plastic is HDPE with a specific gravity of 0.96. The feed waste is 80 ton/h. Find the diameter and the critical speed of the trommel.

Solution:

1. Find diameter of trommel

From Eq. (11),

$$\text{Diameter of trommel } (D) = 7.66\sqrt{\frac{C}{G_\mathrm{s}}}$$

$$D = 7.66\sqrt{\frac{80}{0.96}} = 69.9\,\text{in.} \quad \text{or} \quad 5.83\,\text{ft} \quad \text{or} \quad 1.75\,\text{m.}$$

Trommel diameter is 1.75 m.

2. Find critical speed

   From Eq. (12),

$$v_c = \sqrt{\frac{g}{4\pi^2 r}}$$

$$v_c = \sqrt{\frac{9.81}{4\pi^2\left(\frac{1.75}{2}\right)}} = 0.54\,\text{rotation/s} \quad \text{or} \quad 32.4\,\text{rpm}$$

   Critical speed of trommel is 32.4 rpm.

6.5. *Example 5*

Calculate the air velocity necessary to suspend 2-cm (screened) particles of shredded textile. Note that the drag coefficient of the textile is 1.0, the density of the textile is 0.242 g/cm^3 and the density of the air is 0.0012 g/cm^3. The gravitational acceleration is 980 cm/s^2.

Solution:

From Eq. (18),

$$\text{Air velocity} = v = \left[\frac{4(\rho_s - \rho_a)gd}{3C_D\rho_a}\right]^{1/2}$$

$$v = \left[\frac{4 \times (0.242 - 0.0012)980 \times 2}{3 \times 1.0 \times 0.0012}\right]^{1/2} = 724.16\,\text{cm/s}$$

The required air velocity is 724.16 cm/s.

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Chapter 10

USE OF SELECTIVE RECYCLED MATERIALS IN CIVIL ENGINEERING CONSTRUCTION

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Abstract

It takes a lot of valuable energy and materials to create and manufacture products and the resulting industrial waste can be difficult to manage. The utilization of these industrial waste and by-products can help in reducing the exploitation and consumption of natural resources, reduce greenhouse gas emissions, and assist in the protection of environment. This chapter discusses the properties (physical, chemical, mechanical, and environmental) as well as the techniques for the utilization of fly ash, cement kiln dust, and recycled demolished concrete in civil engineering construction.

Keywords: Recycled materials, engineering construction, industrial by-products, fly ash, types of fly ash, pozzolanic activity, cement kiln dust, soil stabilization, waste solidification, mineral filler, cement substitution, recycled demolished concrete, concrete pavements, road construction, oyster beds.

1. Introduction

All industrial activities generate by-products and waste, hence, waste generation and its disposal form part of the cost of manufacture and act to limit profitability. Waste also causes damage to the environment by polluting air, land, and water.

The gradual diminishing of natural resources and the increasing pollution of the environment makes it necessary to look at the ways for the utilization of the industrial waste and by-products.

The problems associated with the generation and utilization of waste may be approached as follows:

- By changing or modifying the industrial processes to minimize the amount of waste produced.
- By utilizing the waste or by-product generated in the same or other processes or industries, as secondary material or industrial resource.
- By finding methods to neutralize and dispose of the general wastes in a safe and environmentally acceptable manner, in case the utilization of these wastes is not possible.

Though numerous industrial by-products and wastes are being recycled in engineering construction, only the following by-products will be discussed in this chapter.

1. Fly ash,
2. Cement-kiln dust, and
3. Recycled demolished concrete.

2. Fly Ash

Fly ash is a by-product of the burning of finely ground coal in electric power plants. Fly ash typically makes up for 90% of the total ash production. The ash produced by the combustion of coal is recovered in two major forms:

1. The bottom ash is a coarse material similar to sand, which falls to the bottom of the furnace by virtue of its larger particle size and mass.
2. The fly ash is a fine powdery material consisting of glass spheres; some of which are hollow, called cenospheres, and some containing small spheres within large

spheres, called pherospheres. It is collected from flue gases by fabric filtration and or electrostatic precipitation.

The term “fly ash” first appeared in literature in 1937.[1] The use of fly ash in concrete for hydraulic structures and various other structures spread over the United States after the performance data of fly ash used at the Hungry Horse Dam was published by the U.S. Bureau of Reclamation.[2] An ASTM standard on fly ash was established in 1954.[3]

In Japan, studies on fly ash began in 1950 and a standard on fly ash was established in 1958. The utilization of fly ash in European countries came at about the same time. In France, a standard on fly ash cement was approved in 1963, while in the USSR, a standards on fly ash (GOST 6269-63) was approved in 1963. In the UK, a standard on Fly ash (BS. 3892) was approved in 1965.

The world production of coal combustion products (fly ash + bottom ash), according to 2010 estimates, was around 780 million metric tons (Mt). It listed China as having contributed 395 (Mt), North America 118 (Mt), India 105 (Mt), Europe (EU15) 52.6 (Mt), the Russian Federation 26.6 (Mt), Japan 11.1 (Mt) and about 71.7 (Mt) in total by the remaining countries. In 1992, land fills and mine fills were the largest utilization methods contributing to over half of the total world ash utilization; since then, there has been an annual increase in the utilization of fly ash worldwide in various fields of engineering construction. Table 1 lists the production of fly ash in major ash-producing countries around the world in

Table 1. Estimated Annual Production and Utilization of Fly ash in various Countries around the World (2010 Data).

Countries/regions	Total Fly ash and Bottom ash Produced (Mt)*	Total Ash Utilization (Mt)	Total Ash Utilization Rate (%) (Mt)	Total Ash Production/ Person (Mt)	Total Fly Ash Utilization/ Person (Mt)
Australia	13.1	6.0	45.8	0.60	0.27
Canada	6.8	2.3	33.8	0.20	0.07
China	395.0	265	67.1	0.29	0.20
Europe (EU15)	52.6	47.8	90.9	0.11	0.10
India	105.0	14.5	13.8	0.09	0.01
Japan	11.1	10.7	96.4	0.09	0.08
Middle East & Africa	32.2	3.4	10.6	0.02	0.01
USA	118.0	49.7	42.1	0.37	0.16
Other Asia	16.7	11.1	66.5	0.05	0.03
USSR	26.6	5.0	18.8	0.19	0.04

Source: Ref. [5a] ∗ Mt = Million metric tons.

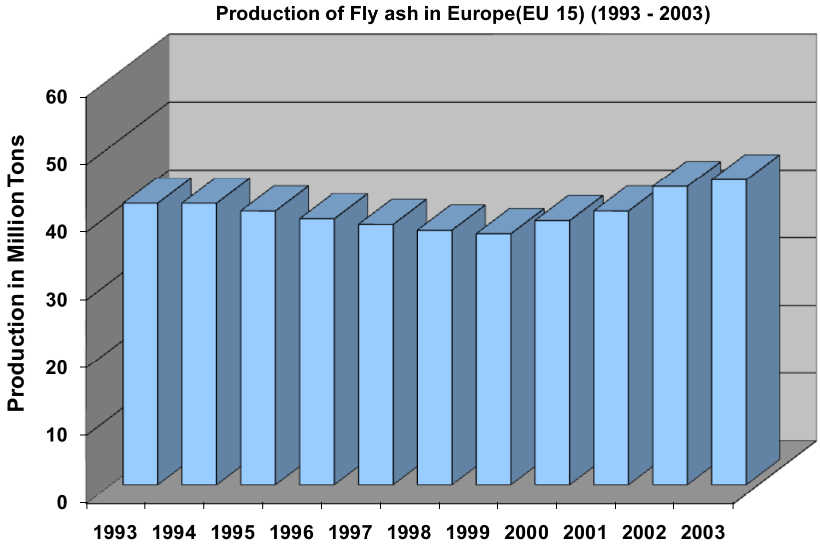

Figure 1. Production of Fly Ash in 15 European Nations from 1993 to 2003 (*Source*: Ref. [5b]).

2010, Fig. 1 depicts the production of fly ash in Europe from 1993 to 2003, and Fig. 2 shows the production of fly ash in 14 European nations percentage-wise, in 1997.

2.1. *Composition and Classification*

Fly ash, which is derived mainly from coal, is made up of a mixture of elemental carbon, complex organic compounds and inorganic substances. When powdered coal is burned, the organic portion is converted to gases such as oxides of carbon and sulfur, and water vapor. Since the inorganic portion is noncombustible, it remains the principle component of the ash. The chemical composition of fly ash depends on the mineral composition (inorganic part) of the coal. The typical fly ash composition produced in various countries is listed in Table 2.

The bulk composition of fly ash, especially silica (SiO_2), alumina (Al_2O_3), ferric oxide (Fe_2O_3), and calcium oxide (CaO), has a significant influence upon its utilization; a high CaO content makes it valuable for cement and soil stabilization, a specific ratio of SiO_2 to Al_2O_3 is important for zeolite synthesis, while a higher sulfite (SO_3) and carbon content prevents its use in concretes.

Magnesium oxide (MgO) can result in the formation of periclase (a crystallization reaction, which results in volume expansion of mineral in concretes), which is undesirable as it results in instability and deterioration of concretes.[7]

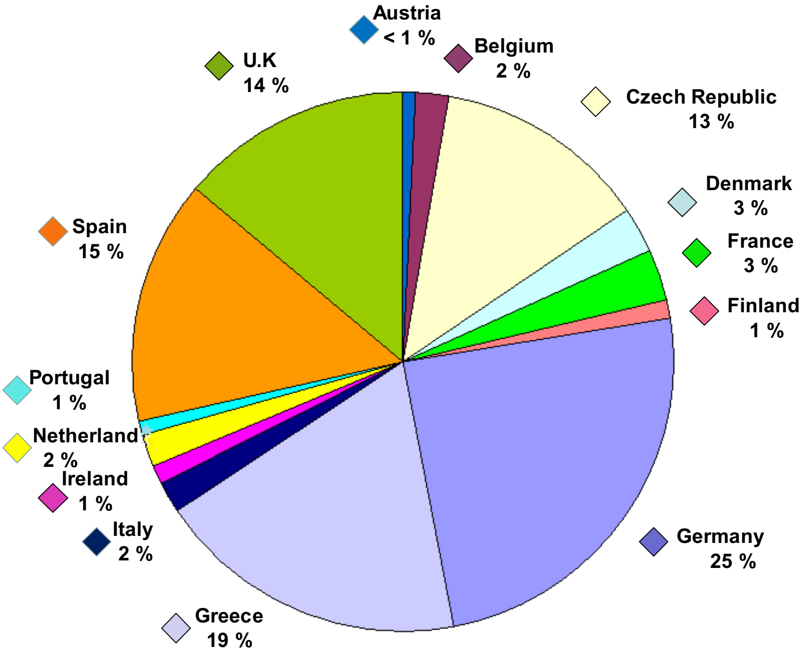

Figure 2. County-Wise Production of Fly Ash in European Nations. The Total Production of Fly Ash of 15 European Nations Approximately 44.0 Million Tons (*Source*: [ECOBA]).

Table 2. Analyses of Class F Fly Ash of Various Countries Around the World.

Country	SiO_2 (%)	Al_2O_3 (%)	Fe_2O_3 (%)	CaO (%)	MgO (%)	Na_2O (%)	K_2O (%)	SO_3 (%)	C (%)	LOI[a] (%)
France	51.68	27.01	6.25	1.72	1.88	0.54	4.49	—	—	4.70
France	48.10	24.68	6.50	1.41	1.82	0.56	4.06			11.70
Belgium	55.74	24.14	6.02	2.47	2.22	—	—	1.04		2.74
Taiwan	48.75	23.21	4.15	3.93	1.00	0.24	1.10	—	—	10.39
The UK	50.09	28.10	11.70	1.62	1.54	0.28	0.62	—	—	1.27
Poland	50.80	23.90	8.60	3.60	2.80	0.80	2.90	0.80	—	2.90
Denmark	53.98	22.27	11.60	3.95	1.97	2.71	Eq.	0.73	—	2.13
The Netherland	50.46	25.74	6.53	4.32	2.24	2.04	4.43	—	—	3.95
Canada	47.00	17.70	25.30	2.10	1.00	0.70	2.30	—	—	2.40
Japan	57.50	26.10	4.00	5.10	1.30	1.50	1.35	0.40	—	1.60
The UK	48.00	38.20	4.50	3.30	1.50	0.30	1.70	0.37	—	2.30
Japan	53.53	23.55	6.23	5.85	1.60	2.20	1.75	—	2.14	3.44
The USA	52.24	19.01	15.71	4.48	0.89	0.82	2.05	1.34	—	0.92
Canada	43.80	22.10	16.20	3.50	0.80	4.40	Eq.	1.10	—	5.00
Germany	51.20	29.60	6.80	3.40	1.20	0.60	3.10	0.50	—	3.30

[a] Often includes carbon content as well. (*Source*: Ref [6])

Sulfite (SO_3) can react with unhydrated and hydrated C_3A to form ettringite (3Cao, Al_2O_3, $3CaSO_4$, $32H_2O$), which has the capability of substituting elements into its structure, making it useful in some waste stabilization applications.[7]

Alkalis (Na_2O and K_2O) react with silicates within aggregates, producing a gel, resulting in swelling and deterioration of concretes; however, fly ash does not contribute reactive alkalis to the concrete and can be added to control these reactions.[7,8] The presence of alkalis is important in the synthesis of geopolymers and zeolite where they play a charge-balancing role as part of the structure.

Loss on ignition (LOI) is a measure of the unburned carbon content of fly ash. Unburned carbon has a large surface area and has a capacity to physically adsorb compounds onto its surface[9]; hence, the carbon content in fly ash is critical because of its adverse effect on air entrainment in concrete. It has been proposed that fly ash carbon, similar to activated carbon, possesses the potential to adsorb surfactants used in air entraining agents[10,11]; therefore, high carbon fly ash or fly ash with varying carbon content must be avoided whenever air entrainment in concrete is a prerequisite. While the quality of fly ash still continues to be monitored by its LOI content, its validity as a predictive tool remains questionable,[12] it has been found that BET surface area seems to give a more realistic view of the carbon content than the LOI.[13]

Elements contained within the glass phase of the fly ash are relatively stable; however, the elements that concentrate within mineral phases or remain at the surface of fly ash usually participate in leaching.

The North American Standard for Fly Ash, ASTM–C618, recognizes two general classes of fly ash, Classes C and F, based upon the type of coal.[14] The most notable chemical difference between these two classes of ash is that Class F contains less than 6% of CaO, designated as low calcium ash, while Class C contains more than 6% of CaO. The basic chemical requirements for the two classes are listed in Table 3.

Table 3. Chemical Requirements of Fly Ash (ASTM C618-93).

Chemical Components	Class C	Class F
$SiO_2 + Al_2O_3 + Fe_2O_3$ (min. %)	50.0	70.0
SO_3 (max. %)	5.0	5.0
Moisture content (max. %)	3.0	3.0
LOI (max. %)[a]	6.0	6.0
Available alkalies (Na_2O eq.) (max. %)[b]	1.5	1.5

Notes: [a]The use of class F fly ash containing up to 12.0% LOI may be approved by the user if either acceptable performance records or laboratory tests are available.
[b] Optional chemical requirements.

2.1.1. *Class F Fly Ash*

This class of fly ash exhibits pozzolanic properties but rarely self-hardening properties. Class F fly ash is normally produced from the burning of anthracite or bituminous coal and with CaO content of less than 6%, it is designated as low calcium ash and is not self-hardening but generally exhibits pozzolanic properties. Class F contains more than 2% of unburned carbon, as determined by LOI test.

In presence of water, Class F fly ash will react with calcium oxide (CaO) or calcium hydroxide ($Ca(OH)_2$) to form cementitious compounds similar to those generated on the hydration of Portland cement.

Because fly ash particles are spherical in shape, Class F fly ash in general improves workability and reduces water demand when mixed with cement or concrete.

2.1.2. *Class C Fly Ash*

This class of fly ash has pozzolanic properties as well as self-cementitious properties since most Class C fly ash contain from 10% to more than 15% CaO. Class C fly ash is normally produced from lignite or sub-bituminous coal and usually contains more than 15% CaO. When mixed with water, Class C fly ash hydrates similarly as Portland cement. The degree of self-hardening depends on the CaO content. The higher the CaO content, the higher the self-cementitious value will be.

The unburned carbon content in Class C fly ash is very low with LOI value being less than 1%.

The glass fraction in fly ash usually varies between 70% and 89% depending on the type and coal source, degree of coal pulverization, combustion conditions in the furnace, and the rate of cooling of combustion residue. It is believed that the calcium in the glassy content of fly ash affects the pozzolanic activity, rather than the free lime.

The European standard for Fly ash, BS EN 450, also classifies fly ash in two general classes; siliceous fly ash (V), and calcareous fly ash (W). Siliceous fly ash must contain less than 5% (by mass) of reactive CaO and at least 25% (by mass) of reactive silica, while in calcareous fly ash the content of reactive CaO must not be less than 5% by mass. Calcareous fly ash, containing between 5% and 15% by mass of reactive CaO, must contain more than 25% by mass of reactive silica.

The EN 450–1994 has been divided in to two parts, EN 450-1 deals with definitions, specifications and conformity criteria, whereas EN 450-2 covers the conformity evaluation of fly ash for concrete. The standard includes a regular factory production control, internal quality control and auto control testing of samples, an audit testing of samples, an inspection of the production plant and production control by a notified certification body. The established system resembles the certification system

Table 4A. LOI Categories of the Revised EN-450.

	LOI in % by Mass	
Category	Min.	Max.
A	—	5.0
B	2.0	7.0
C	4.0	9.0

Source: Ref. [5].

Table 4B. Types of Co-combustion Materials of the Revised EN-450-1.

1	Vegetable materials such as wood chips, straw, olives shells, and other vegetable fibers
2	Green wood and cultivated biomass
3	Animal meal
4	Municipal sewage sludge
5	Paper sludge
6	Petroleum coke
7	Virtually ash free liquid as gaseous fuels

for cement very much. The fly ash producer (either the power plant or marketing company) is responsible for the quality, efficiency and reliability of the product.

In 2005, the revised EN 450 included some fundamental amendments; the most important ones from the viewpoint of the producers are:

- Introduction of three categories of LOI (Table 4A);
- Introduction of two classes of fineness;
- The acceptance of fly ash obtained from the co-combustion of certain types of biomass and waste (Table 4B); and
- Acceptance of fly ash processed e.g. by classification, selection, sieving, drying, blending, grinding, carbon reduction or by a combination of these processes in an adequate production plant as fly ash, according to the standard.

The standard refers to siliceous fly ash only. Calcareous fly ash, as obtained mostly from the combustion of lignite, cannot be utilized as concrete addition according to the EN 450. This fly ash is used mostly for the filling of opencast mines. To increase the use of fly ash, the European Coal Combustion Products Association (ECOBA) is active in the development of European standards for fly ash, especially the EN 450 "Fly ash for Concrete" and pr-EN 14227 "Hydraulically bound mixtures — Specification — Part 3: Fly ash bound mixtures" and "Part 4 : Fly ash for hydraulically bound mixtures."

2.2. *Fly Ash Properties*

2.2.1. *Fineness*

ASTM C-618 requires maintaining particle size of both the classes of fly ash, so that the amount retained on a 45-μm sieve should not be more than 34%.[14] It is well known that the particles larger than 45-μm size show little or no pozzolanic reactivity under normal hydration conditions during the first 28 days. The pozzolanic activity of particles under 10-μm size, however, was found to be directly proportional to the amount of particles present in the reaction.[15]

2.2.2. *Density*

Since the density of fly ash is related to its morphology, it could also be a determining factor of its pozzolanic activity. Fly ash is composed mainly of cenospheres (hollow spheres with skin of mainly reactive glassy or amorphous silicon dioxide) and sometimes of pherospheres (many small hollow spheres contained in bigger hollow spheres).

Gases from the coal combustion process bloat the semi fluid particles into spherical shape. The density of spheres depends on their size; the smaller the size is, the higher the density will be. The glassy hollow spheres, regardless of their size, tend to reduce the density of the ash (as measured by standard methods). Blaine apparatus, commonly used for surface area determination of Portland cement as an easy substitute for particle size distribution, is not suitable for very fine and microscopic materials. With such materials, another method called BET adsorption is preferred. From a study of U.S. fly ashes,[15] it was reported that typically more than 40% of the particles were under 10 μm and less than 15% were above 45 μm.

2.2.3. *Carbon Content*

The LOI is the measurement of unburned carbon remaining in the ash. The unburned carbon can absorb air-entraining admixtures (AEA) and increase water requirement in air-entrained concrete. For some of the fly ashes, there is significant difference in carbon content and LOI, the specifications, however, restricts the LOI and not the unburned carbon. The ASTM C 618 restricts the LOI up to 6% while AASHTO sets the limit to 5%. Variations in LOI can contribute to fluctuations in entrained air in concrete. There is also a disadvantage that, due to the high carbon content in fly ash, the carbon particles may float to the top during the concrete finishing process producing dark streaks on the surface.

2.3. *Pozzolanic Activity*

The pozzolanic activity of fly ash generally refers to its ability to react with calcium hydroxide ($Ca(OH)_2$) and form hydrates at ambient temperature. The reactivity of pozzolana with CaO has been studied[16–18] and is found to be influenced by its inherent characteristics such as chemical and mineralogical positions, morphology, the amount of glassy phase, the unburned carbon content or LOI and the fineness of the fly ash.[19]

Several investigators have reported that when fly ash is pulverized to increase fineness, its pozzolanic activity increases significantly. However, the effect of increase in Blaine surface area beyond 6,000 cm^2/gm is reported to be insignificant.[20] It is also influenced by external factors such as thermal treatment[21] and the addition of admixtures.[22–24]

The two important parameters indicative of the reactivity are its CaO and carbon content. Class C fly ash with high CaO content exhibits self-hardening properties besides pozzolanic activity. Some Class C fly ashes generate heat of hydration immediately on adding water.

Carbon content in fly ash is harmful to its activation and acts as a diluent of the active pozzolanic matter in fly ash. Different organizations, therefore, put a limit on LOI in fly ash from 3 to 6%. LOI even in excess of 0.5% seems to significantly affect the pozzolanic activity as well as some other performance characteristics in Class C fly ash.[25]

The term "pozzolanic activity" covers all reactions occurring among the active constituents of pozzolana, lime, and water. It includes two parameters, the maximum amount of lime that a pozzolana can combine and the rate at which such combination occurs; both factors depend on the quality and quantity of active phases. After 180 days, pozzolana has been able to combine 50–100% of lime with respect to their mass. In pastes, lime combination is lower, since the hardened mass hinders the attainment of equilibrium conditions.[26]

Researchers generally agree that the overall amount of combined lime essentially depends on the nature of active phases, their content in pozzolan, silica content, lime: pozzolan ratio, length of curing, BET surface area of pozzolan, the water/solids ratio, and temperature.[27]

Palomo *et al.*[22] found that when the alkali solution contains soluble silicates (sodium or potassium silicate) the reactions occur at faster rates than when hydroxides (sodium or potassium hydroxides) were used as activators. The reaction steps were found to be so fast that the dissolution of chemical species, the accumulation of reaction products, and the polycondensation of structures occur almost simultaneously, consequently the development of mechanical strength is quicker in the first case (samples activated with soluble silicates) than in the second case (samples activated with the hydroxides).

Lee *et al.*[28] found that the addition of activators such as sodium sulfate, potassium sulfate, and triethanolamine accelerated the strength development of the samples at early ages, but their strengths at 28 days were comparable to strengths of fly ash mortar samples without activators. Of these activators, potassium sulfate was the most effective in increasing the early strength of the fly ash.

Alexander[24] has shown that the strength development of a porous opaline pozzolana–lime mortar can be improved by the addition of sodium hydroxides. The lime–pozzolan reaction is accelerated by increasing the temperature resulting in the gain in the strength development. Above 70°C, however, combined lime tends to stop increasing or decreasing the reaction — likely due to the variation in the composition of the hydrated phases.[29]

2.4. *Utilization of Fly Ash*

Fly ash must be viewed as a commodity with a promising economic future and as a valuable raw material rather than an unwanted solid waste. The potential for fly ash utilization is an excellent illustration of how an industrial by-product can be successfully reintegrated into a nation's economy.

The utilization percentage of fly ash in most of the European nations is from 60% to as much as 99% of their total fly ash production, while in the USA, the utilization percentage is around 35.3%; and over the nine years from 1992 to 2001, the proportion of ash utilization increased by only 8% while the total fly ash production increased by 50%. In Asia, the major consumers of fly ash are Korea at 93.6% followed by Japan at 52.1% and China at 37.4%.[6]

The percentage utilization of fly ash in various sectors of civil construction worldwide is illustrated in Fig. 3. Of the total fly ash production, about 45% is utilized in the manufacture of cement, concrete, concrete products and in road construction. While the rate of utilization of fly ash in the European nations varies widely, the overall level of utilization, however, is significantly higher than in other countries around the world (Fig. 4).

The pattern of fly ash utilization in the USA in 1999 is presented in Table 5. Of the total production of coal fly ash, 67% were sent to landfills for disposal.

The data from Canada (2001) showed a 22% utilization of fly ash for cement, concrete/grout and mining applications and the cumulative use percentage was 27%. In Australia, the total ash production in 2003 was 13 million tons. About 2.5 million tons were used for cementing applications, while in Japan, about 9.5 million tons of coal ash was produced and about 8.5 million tons were beneficially utilized, mostly in the cement industry.

Although the characteristics and composition of fly ash may vary in physical and chemical properties, it is widely accepted that fly ash disposal should be avoided whenever possible in order to protect the environment, therefore, expanding the utilization of fly ash is an important challenge. Figure 5 describes the various

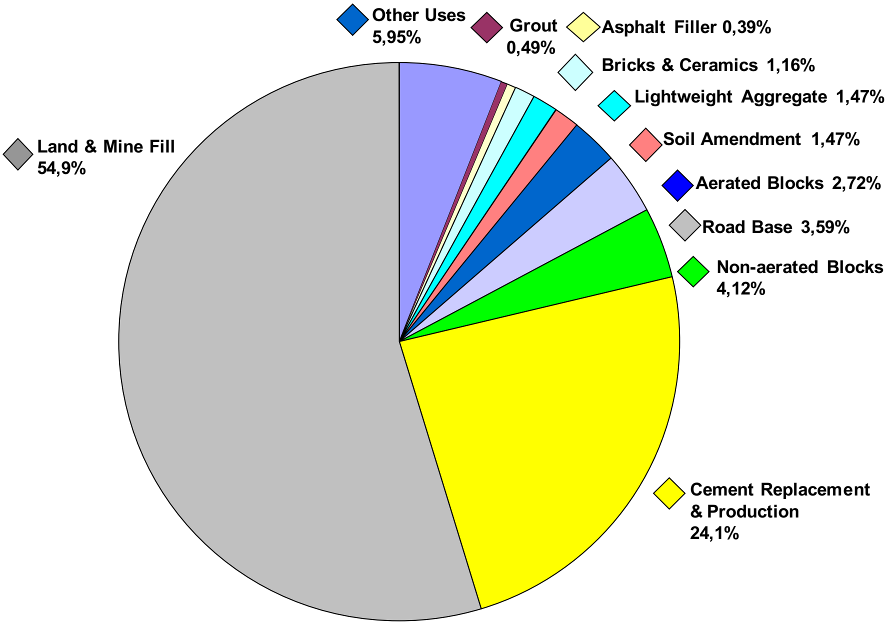

Figure 3. Utilization of Fly Ash in Various Fields of Engineering Construction Worldwide (*Source*: Ref. [6]).

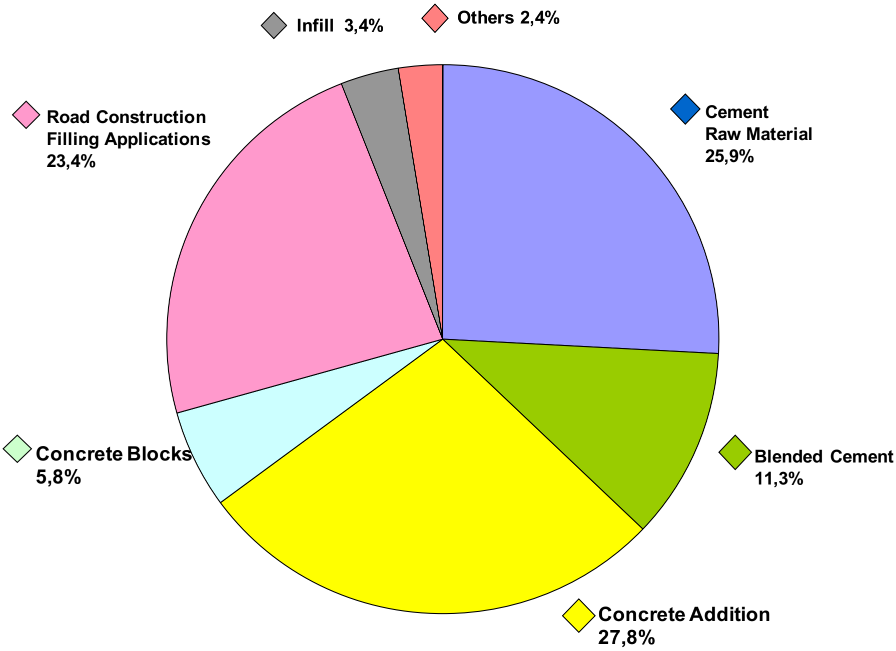

Figure 4. Utilization of Flyash in Various Fields of Engineering Construction Europe. Total Utilization 21.1 Million Tons (*Source*: Ref. [5b]).

Table 5. The Patterns of Coal Fly Ash Utilization in the USA in 1999.

Utilization	Million Tons/Annum (%)
Cement/concrete/grout	10.3
Flowable fill	0.8
Structural fill	2.9
Road base/sub-base	1.1
Mineral filler	0.1
Mining applications	1.4
Waste stabilization/solidification	1.8
Agriculture	0.1
Miscellaneous/other	0.4
Total use	18.9
Percentage usage	33.2

Source: ACAA, April 2001.

utilization strategies for fly ash, keeping in view the variations in the quality of fly ash.

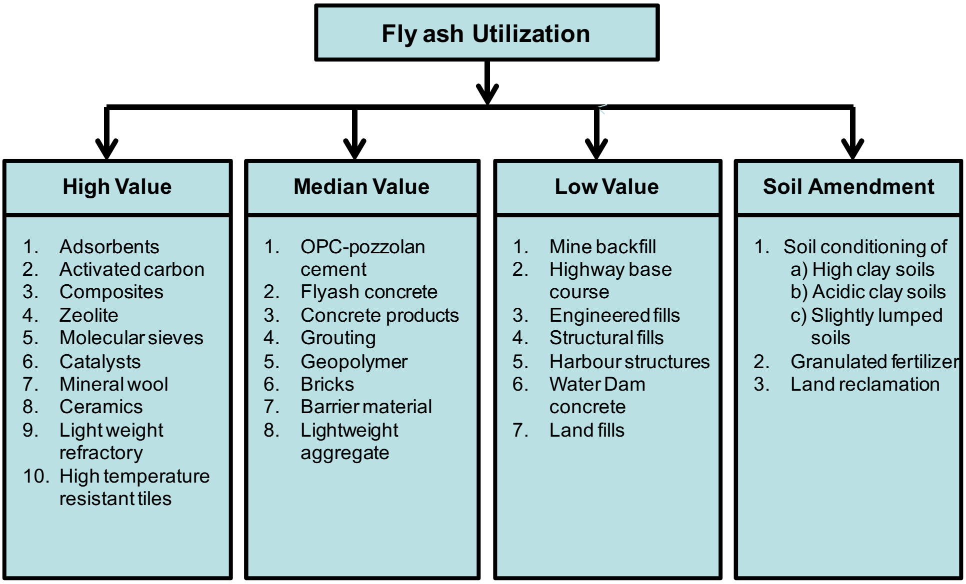

Figure 5. Fly Ash Utilization Strategies (*Source*: Refs. [31, 121]).

2.4.1. *Cement and Concrete Production*

There is great potential for the use of fly ash in the construction industry due to its pozzolanic properties. Fly ash can be incorporated into nearly every stage of the production and use of cement. Fly ash contains lime, silica, alumina, and ferric oxide that are essential components needed in the manufacture of cement; therefore, it can be used as one of the components of cement raw mix.

Raw mix is added to the kiln and heated to temperatures of 1450°C; the raw materials melt, react with each other, and the sintered material emerge from the kiln as large round-shaped granules called clinker, which is ground together with small percentage of gypsum into a fine powder of required fineness — the cement.

Fly ash can be added at a required ratio during cement grinding to make Portland–pozzolan cement or low heat cement — or it can be added latter as a partial replacement of cement in concrete.

Cement, when hydrated, produces high heat of hydration that creates shrinkage cracks on the concrete surface, but the addition of fly ash to cement reduces its heat of hydration preventing surface cracks; it is, therefore, ideal for use in mass concrete projects such as large pipes, large precast construction components and dams.

Moreover, the addition of fly ash to concrete increases its density, reduces its permeability and makes it much more durable; hence, fly ash addition is beneficial in the production of sewerage pipes, since low permeability will increase resistance to the ingress of sulfate and weak acids that are usually found in sewer lines.

Another useful product that can include fly ash as a component is lightweight concrete that contains a combination of ingredients that cause the concrete to foam. The lightweight concrete is only 25% the weight of regular concrete but has superior insulating properties.

2.4.2. *Waste Solidification and Soil Stabilization*

Waste solidification and soil stabilization are done by the alteration of waste or soil properties to improve the engineering properties of soils e.g. the compressive and shear strength of the waste or soil, to control the shrink and swell of plastic soil and reduce the water content of wet soil.

In addition to affecting the physical properties of waste or soil, fly ash can be used to neutralize acidic soil to prevent the leaching of contaminants through these soils and to bind contaminants within the waste or soil.

The types of clay minerals present in the soil and their adsorbed water influence the plasticity of soils treated with high calcium fly ash. Organic soils and soils containing more than 10% sulfates are difficult to stabilize with the addition of fly ash.

The plasticity index as determined by ASTM D 4318 is limited to 10–12 to ensure a stable material. The higher the plasticity index, the higher the potential of the soil to shrink or swell as it undergoes moisture content fluctuations. To lower the plasticity of soils, historically, plastic soils have been treated with quick lime (CaO) or hydrated lime ($CaOH_2$). The lime chemically reacts with the soil particles and changes the soil grain from clay size (less than 0.002 mm) to silt size (0.05–0.002). The aluminosilicates contained in fly ash dissolve in the high pH of lime and produce a cementitious compound, which controls shrink or swell by cementing the soil grains and restricting the movement of soil particles. Typically, 12–17% of fly ash (based on the dry weight of soil) is added.

The swell potential of fly ash treated soils under confining pressures of 48 kPa is less than 0.5%, even when compacted 2–4% below optimum moisture content for maximum density.

2.4.3. *Flowable Fill*

Flowable fill is a mixture of water, fly ash, Portland cement and coarse or fine aggregates (or both) with flowable consistency. It is an engineered, strength controlled fill material that is self-leveling, nonsettling and self-compacting. Flowable fills are proportioned, mixed and delivered in very workable (flowing) concrete and provide for an in-place product equivalent to high-quality compacted soil without the use of compaction equipment and related labor. Flowable fill is used to save on time — as compared to other fill materials that need compaction and take up to six times longer for placing and compacting.

Flowable fill can be used to backfill trenches, around pipes, under roadways, abutments, and retaining walls. It can also be used to fill abandoned underground tanks, pipelines and utility vaults because it is self-leveling and leaves no air spaces.

Flowable fill also has the advantage that it can be utilized even in cold temperatures of winter. If the material does freeze and breakdown, it essentially becomes well-graded fill and can be left as it is.

Virtually any coal fly ash can be used in flowable fill — even fly ash with high carbon or LOI content — in a dry or moisture conditioned form is suitable. The fly ash does not have to meet ASTM C 618 specification requirements to be suitable for use in flowable fills.

Low fly ash mixes that contain higher percentages of fine aggregate has been designated as controlled low-strength material (CLSM) and has an upper compressive strength limit of 8,300 kPa. Most flowable fill mixes are designed to achieve a maximum strength of 1000–1400 kPa so as to allow for excavation at a later time. The high fly ash mixes normally contain 95% fly ash and 5% Portland cement.

2.4.4. *Mineral Filler in Asphalt Paving*

Fly ash has been successfully used as a cost effective mineral filler in hot mix asphalt. Mineral fillers increase the stiffness of the asphalt mortar mix, improving the rutting resistance of pavements, and help reduce the amount of asphalt drain-down in the mix during construction, improving the durability of the mix by maintaining the amount of asphalt initially used in the mix.

Current specifications for mineral filler in hot mix asphalt, i.e. AASHTO M 17; require the material passing to be between 70% and 100% on No. 200 mesh sieve. The particle size of most fly ashes range between 60% and 90% passing through a No. 200 mesh sieve. Fly ash generally can meet mineral filler specification requirements for organic impurities and plasticity. Owing to its hydrophobic and alkaline nature, fly ash reduces the potential for stripping of asphalt.

Asphalt pavements with coarse gradations are increasingly being designed since they perform well under heavy traffic conditions.

2.4.5. *Structural Fill*

The use of fly ash as a structural fill to construct fills and embankments is the second most common beneficial application. Specifications for fly ash structural fills and embankments are similar to specifications for engineered soil fills. To achieve proper strength and compressibility characteristics for supporting roads and structures the fly ash fills are required to be placed properly and compacted at optimum moisture content. Class F fly ash is generally used in structural fills because the addition of moisture is necessary for its construction — through a Class C fly ash that is high in calcium oxide can pose a handling problem and potential inability to achieve the required degree of compaction.

Two types of specifications are typically applied for the construction of fly ash embankments: performance specifications and method specifications.

1. *Performance specifications*: Performance specifications designate the allowable moisture content and the required degree of compaction. For road embankment, the fly ash is compacted to 95–100% of maximum dry density as determined by ASTM D 698 (AASHTO T 99). The allowable range for the moisture content is determined by plotting the laboratory moisture–density relationship.
2. *Method specifications*: On some projects method specifications are preferable over performance specifications. The method specification is based on the findings of field compaction tests on trial strips. To achieve the necessary degree of compaction, the required lift thickness, weight of compaction equipment, its speed, and the number of passes are specified. Since the compaction procedures can be monitored visually, the quality control of the method specification is simpler.

3. Cement Kiln Dust

Cement kiln dust is a partially calcined high alkaline by-product of Portland cement manufacturing and is, therefore, sometimes considered as industrial waste. In 1999,

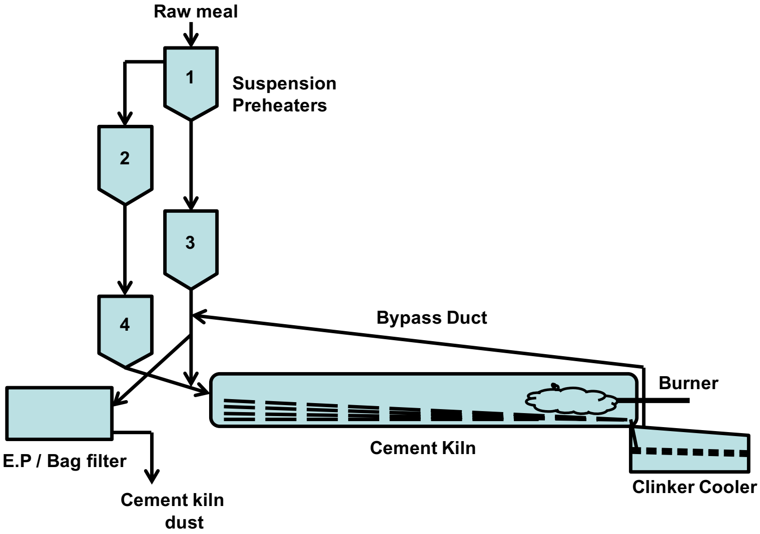

Figure 6. Portland Cement Manufacturing Operations.

it was estimated that 30 million tons of cement kiln dust were produced worldwide[32] of which approximately 15 million tons of cement kiln dust was produced annually by the American Cement Industry.[33] Due to its high alkaline content, most of the material was disposed of on-site without any further reuse.[34–36]

In the manufacture of Portland cement, a mixture of limestone, shale, clay and sand (raw materials) are combined in controlled proportions, ground to fine particle size (raw meal) and either fed to the rotary cement kiln through suspension preheaters (dry process), or in the slurry form fed from slurry basins, (wet process) (Fig. 6). The kiln charge reaches sintering temperatures and the sintered material is converted to round-shaped granules, called clinker, due to the circular motion of the kiln. With it calcined and partially calcined dust is also formed in the kiln, which is carried from the cement kiln by the exhaust gas and collected by the kiln's air pollution control devices such as the electrostatic precipitator, bag-house filter or multi-cyclone. This dust contains a mixture of partially calcined clinker, uncalcined feed materials, fuel combustion by-products and alkali compounds.

The specific chemical and physical characteristics of the cement kiln dust depend on the characteristics of the raw materials, fuel, kiln process and the method used for dust collection. The X-ray diffraction analysis of kiln dust indicates that it consists mainly of lime stone as the main component and minor amounts of quartz together with calcium sulfate ($CaSO_4$), common salt (NaCl), potassium sulfate (K_2SO_4), spurite ($2C_2S$, $CaCO_3$) and sulfo-spurite ($2C_2S$, $CaSO_4$).[37]

Free lime can be found in cement kiln dust and its concentration is typically higher in the coarser particles. Finer particles tend to exhibit higher concentrations of sulfates and alkalis. Cement kiln dust collected from wet-process kilns tend to have lower calcium content than kiln dust from dry-process kilns.

3.1. *Material Properties*

3.1.1. *Chemical Properties*

The chemical composition of cement kiln dust is similar to conventional Portland cement; its principal chemical constituents are calcium carbonate, lime, silica, crystalline silica, alumina, ferric oxide, potassium oxide, sodium oxide, sulfur trioxide, and magnesium oxide. Table 6 lists the typical composition of cement kiln dust.

There is little, if any, free lime or free magnesia content in stockpiled cement kiln dust that has been exposed to the environment for long periods.[38] Due to the alkaline nature of cement kiln dust, with $pH = 12$, it can corrode some metals that come in direct contact with it.

Under the U.S. Environmental Protection Agency's (EPA) resource conservation and recovery act (RCRA) regulations, cement kiln dust is not considered to be hazardous material. However, this does not mean that cement kiln dust does not contain anything that could pose a hazard to the environment; however, the toxicity of cement kiln dust must be determined on a case-by-case basis.[39] Trace elements in cement kiln dust e.g. cadmium, lead and selenium are generally found in concentrations less than 0.5% by weight. Because some of these constituents are potentially toxic at low concentrations, it is wise to assess their levels (or leachability) before considering its use.[40] In order for cement kiln dust to remain a safe material, it has to be handled properly to prevent environmental contamination.

Table 6. Typical Chemical Composition of Cement Kiln Dust.

	Cement Kiln Dust		
Parameter	Fresh[a]	Stockpiled[a]	German[b]
CaO	40.5	31.4	34.40
SiO_2	14.5	11.7	15.90
Al_2O_3	4.10	3.18	3.78
Fe_2O_3	2.00	2.16	2.58
Na_2O	0.44	0.13	0.41
SO_3	6.50	8.24	7.74
MgO	1.55	0.97	0.78
LOI	22.9	40.4	23.95
Free lime	4.4	0.0	—

Source: [a]Ref. [38], [b]Ref. [41].

Table 7. Typical Range of the Physical Properties of Cement Kiln Dust.

Properties	Cement Kiln Dust
Physical state	Solid (powder)
Appearance/odor	Gray to tan powder/odorless
pH (in water) (ASTM D 1293-95)	12–13
Specific gravity	2.6–2.8
Specific surface	4600–14000
Maximum particle size	0.300 mm (No. 50 sieve)
Gradation 75% passing	0.030 mm (No. 450 sieve)
Solubility in water	Slightly soluble (0.1–1.0%)

Source: Ref. [37].

3.1.2. *Physical Properties*

Cement kiln dust is a fine powdery material; its fineness covers a wide spectrum, i.e. from 4600 to 14000 Blaine surface areas. Typical range of the physical properties of cement kiln dust is listed in Table 7.

Since cement kiln dust is partially calcined, its specific gravity falls in the range of 2.6–2.8, less than Portland cement (specific gravity 3.15–3.20). To prevent the loss of free lime contained in cement kiln dust and to maximize its reactivity, it should be handled like normal cement.

The total amount of calcium oxide present in the kiln dust, which includes both free and bound limes, varies from plant to plant due to variation in raw materials and pyroprocessing. The LOI is an important indicator of free lime present in the dust.

A higher LOI indicates that the dust sample contains a higher percentage of bound water within its chemical structure and has less calcium oxide available for hydration reactions. A lower LOI means less bound water and more free lime available for hydration reactions.[41]

Though cement kiln dust contains many of the same materials as cement, its alkali, sulfate and chloride content can cause adverse effects on concrete quality. The alkali content of cement is required to be less or equal to 0.6% Na_2O equivalent (Na_2O + 0.66 K_2O) by the ASTM C-618, the Canadian CSA A3001-A 23.5, and cement standards of most of the other countries. Higher percentage of alkali contained in the cement kiln dust may have an increased potential for alkali–aggregate reactions that reduce the durability of concrete significantly.

3.1.3. *Mechanical Properties*

Cement kiln dust has a loose density of approximately 480 kg/m^3. When compacted using conventional soil compaction methods, it acquires the density of 1350–1500 kg/m^3.[42]

3.1.4. *Environmental Properties*

Cement kiln dust is generally considered nonhazardous and has relatively low potential for leaching heavy metals. Trace constituents in cement kiln dust include cadmium, lead selenium and radionuclides which are generally found at concentrations of less than 0.05% by weight. Since these constituents are toxic, it is important to assess the mobility and leachability of these trace contaminants in the dust.

A solid waste is considered to be hazardous if it exhibits ignitability, corrosivity, reactivity or toxicity. It has been determined using EPA procedures, that cement kiln dust meets the standards for the first three criteria. The toxicity of cement kiln dust must be determined on a case-by-case basis and the tested materials cannot have concentrations of arsenic, barium, cadmium, chromium, lead, mercury, selenium or silver over the allowable limits.[43]

In recent years, dust recirculation systems in the cement kilns have become more common and in certain cases almost all cement kiln dust is recycled through the kiln with the raw materials to make cement. As a result, the cement kiln dust gradually becomes more and more concentrated with contaminants with each cycle through the system.

While the recirculation of cement kiln dust is the primary factor in its contamination, the use of alternative fuels (rubber tires, spent solvents, tars and bitumens, tallow etc.) as fuel and raw material substitution with alternative raw materials (wastes rich in iron, silica, alumina or calcium carbonate) is another equally important issue. Therefore, there is concern that heavy metals such as cadmium, chromium and lead from the waste fuel and alternative raw materials could accumulate in the cement kiln dust or even adsorbed on the surface of the clinker and, therefore, in the cement itself.

Sprung and Rechenberg[44] established lead and zinc balances for raw feed, fuel, clinker and kiln dust. From the material balances, it appears that 86% of the zinc is combined in the clinker with the remainder in the dust, whereas only half of the lead is combined in the clinker and the other half in the dust.

The concentrations of various elements in the kiln dust and kiln dust solutions were examined by Duchesne and Reardon.[45] They found that many of the phases of cement kiln dust such as lime (CaO), arcanite (K_2SO_4) and sylvite (KCl) are unstable or highly soluble in water or will precipitate in the form of less soluble secondary phases during the initial hydration reactions between cement kiln dust and water; Fig. 7 shows the process of kiln dust collection and its treatment and handing.

Major components of cement kiln dust (mainly potassium, sodium, carbonate, sulfate and chloride) are very soluble and have high leachate concentrations; over 40–50% of cement kiln dust by weight dissolves in water. Potassium, sodium and sulfates are the main components that dissolve into the solution at such a high rate.

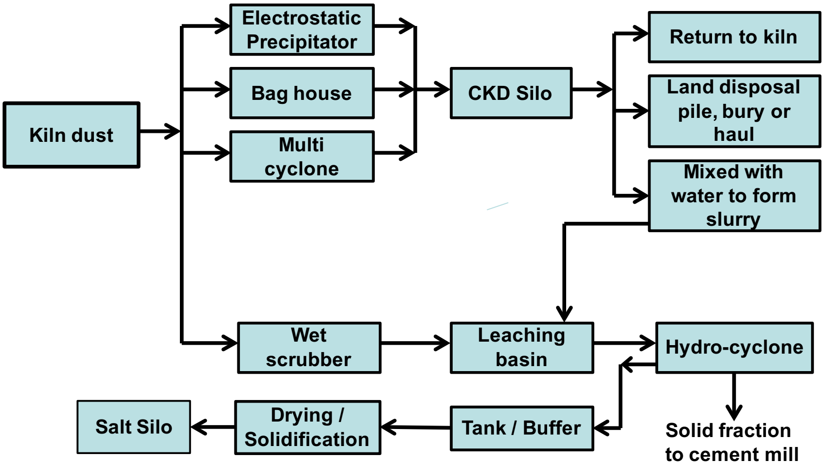

Figure 7. Kiln Dust Collection and Handling (Adopted from Ref. [58]).

Often the two materials are found in leachate as potassium hydroxide and potassium sulfate.[45]

3.2. *Beneficial Uses*

The primary value of cement kiln dust is its cementitious properties since its chemical composition is similar to that of cement. Its particle size and alkalinity also provide value for a variety of beneficial uses. The construction, waste treatment and agricultural industries all stand to benefit from the reuse of cement kiln dust. Some of the potential beneficial uses of cement kiln dust are discussed in the following sections.

3.2.1. *Soil Stabilization*

Soil stabilization is becoming increasingly important in road construction, and cement kiln dust is probably one of the most extensively used material as a binder in soil stabilised base and sub-base pavement applications. In the USA, 10% of the total usage of the cement kiln dust is for soil stabilization. As an alternative to lime, which is sometimes used in road construction, cement kiln dust can be used for the stabilization of soil. When lime is used, the soil must be remixed and compacted 48 h after the lime is first applied, but with cement kiln dust the initial mixing and compacting of soil is enough to stabilize the material sufficiently.[41] Therefore, the utilization of cement kiln dust not only improves soil strength, but also economizes the cost and working time.

Previous research has shown that cement kiln dust is a potentially useful soil stabilizer. Baghdadi[46] found that the unconfined compressive strength (UCS) of kaolinite samples mixed with 16% of cement kiln dust, increased from 210 to 1115 kPa when cured for 28 days. An increase of California Bearing ratio (CBR) from 2% to 30% was experienced by Soomro[47] with samples of fine-grained soils treated with 17% fly ash and 7.5% cement kiln dust and cured for seven days. Miller and Azad[48] in their study found that 20% addition of cement kiln dust increased the 28-day UCS from 250 to 1800 kPa for soils with plasticity index of 33 and 40, while a third type of soil with plasticity index of 6 gave 3500 kPa UCS at 28 days. The available literature indicates that cement kiln dust not only effectively improves soil strength, but also reduces construction time and costs.

3.2.2. *Waste Solidification*

Another of the most common uses of cement kiln dust is for the stabilization and solidification of waste. Wastes from industrial wastewater, sewage, oil sludge,[49] and coalmine effluent[50] have been stabilized using cement kiln dust. The adsorptive quality of the dust and its alkaline nature allow it to reduce moisture content and increase the bearing capacity of waste and neutralize it, immobilize hazardous constituents and control residual odor.[51] Cement kiln dust is a cost-effective alternative to other conventional waste treatment materials such as lime and cement.

3.2.3. *Asphalt Concrete Mineral Filler*

Cement kiln dust has also been widely used as mineral filler in asphalt concrete mixes. The blending of cement kiln dust into the hot mixed asphalt cement binder prior to incorporation with the hot mix aggregate, results in a binder that can reduce significantly the asphalt cement volume requirements by 15–25%. In addition to its desirable physical properties, the cementitious properties of cement kiln dust have been found to increase the stability and stiffening of hot mixed asphalt.[52] The less ductile mastic asphalt made with the incorporation of cement kiln dust has been successfully used as a waterproofing and protection agent.

3.2.4. *Cement Replacement Material*

Cement kiln dust has similar pozzolanic properties and physical characteristics as Portland cement. The high alkali and sulfate content makes it an excellent activator for materials with pozzolanic properties. Due to its finer particle size, it can be used as a potential substitute for cement in the manufacture of non-load-bearing concrete products such as concrete blocks, pavers, roof tiles and bricks.

Replacement of pure Portland cement with 5% cement kiln dust (CKD) showed no significant difference in the concrete's compressive strength. The performance of concrete with 5% CKD was almost similar to that of concrete without CKD. A similar trend was noted in the drying shrinkage. The chloride permeability and electrical resistivity data indicated that the chances of reinforcement corrosion would increase with 10% and 15% cement kiln dust.[53] In another study, substitution of 5% cement kiln dust did not show any appreciable adverse effects on the development of compressive strength at water to binder ratio = 0.5. The reductions in 28-day compressive strengths at the substitution rates of 5% and 10% cement kiln dust were found to be 1.8% and 4.5%, respectively.[54]

Studies were also carried out on the activation of ground granulated blast furnace slag and fly ash with the addition of cement kiln dust to produce nonconventional cementitious binders for concrete. The study on the heat of hydration of slag activated with 50% cement kiln dust revealed that the trend of heat evolution was similar to that of Portland cement (OPC)–slag blend.[55] When OPC is hydrated, ($Ca(OH)_2$) is formed as one of the hydration products. The high pH of ($Ca(OH)_2$) activates slag (secondary activation) and ($Ca(OH)_2$) is thus consumed. Similarly in the above study, it was also found true that no ($Ca(OH)_2$) was detected after 7-days hydration indicating that all the available ($Ca(OH)_2$) was consumed in the activation of slag. Due to the release of sufficient sulfate ions reacting with ($Al(OH)_4$) well crystallized Ettringite ($3CaO$, Al_2O_3, $3CaSO_4$, $32H_2O$) was formed.[55] In another study on the effect of (NaOH) and curing temperature on the hydration of CKD and Fly ash blend, the major crystalline hydration product of CKD — Fly ash blend was Ettringite as well. When 2% sodium hydroxide was added to the blend, it accelerated chemical dissolution but depressed crystal formation of both Ettringite and calcium hydroxide during hydration. Curing at elevated temperatures had a more significant influence on the cement kiln dust–fly ash binder strength development compared to Portland cement strength development and appeared more effective in strength improvement than sodium hydroxide additives. Binder made with 50% cement kiln dust, 50% fly ash blend and 2% sodium hydroxide, and cured at 38°C gave satisfactory compressive strength.[56]

3.2.5. *Production of CLSMs*

CLSMs, also known as “flowable fill”, is mainly used for filling cavities in civil engineering works in which the application of granular fill is either impossible or difficult. The cohesiveness of highly flowable concrete-like material is generally poor, leading to the risk of segregation unless a high amount of fines are used in the mix. The cementitious properties of cement kiln dust make it a viable and cheap substitute for the replacement of a portion of Portland cement in flowable fill mixtures.

In a laboratory study based on mixtures of cement kiln dust:fly ash ratios of 1:12, 1:6, and 1:1 and varying water to binder ratios from 0.95 to 1.10, flowability, bleeding, setting times, mass density and 28-day UCS were analyzed. Test results revealed that high flowability and setting times could be achieved with most mixtures within 24 h. Although bleeding levels were found to be higher than normal, the strengths of up to 460 kPa were achieved. Hence, it was concluded that cement kiln dust could be beneficially added to produce very low-strength material that offers comparable strengths to soils used for conventional fills and many other low-strength applications.[57]

3.2.6. *Agricultural Land Application*

Because of high potassium and lime concentrations, CKD is used as soil amendment or fertilizer in many parts of the world. A high percentage of soluble potassium provides a good source for growing plants and lime in turn provides calcium to soil. In a study done on potato crops in Quebec, Lafond and Simard[58] found that while cement kiln dust added more potassium and calcium to the soil, there was no crop contamination by heavy metals. The properties of cement kiln dust make it an effective and safe agricultural additive, particularly for acidic soils.

4. Recycled Demolished Concrete

Besides the use of industrial by-products, the demolition of concrete structures such as airfield pavements and buildings is also a source of materials to be reused in new engineering applications. More than 200 million tons of recycled aggregates are generated each year in the USA.[59]

Solid waste has been an inevitable by-product of the operations of industrialized societies. One result of economic growth is an increase in generation of solid waste. This combined with the decreasing number of available or suitable sites has led to a global reassessment of the methods employed to dispose of solid waste. Solid waste disposal is not simply a physical problem; its transport contributes to fossil fuel usage and the emission of green house gases.

Some studies have estimated that up to 90% of demolition waste going to landfills can be recycled and reused. However, such studies need to be tested in a more comprehensive manner with indicative analysis of the actual practice.

Very large quantities of broken concrete are produced as waste material in the building, construction and demolition industries. The depletion of the supply of quality aggregates together with environmental, economic and energy considerations have encouraged the recycling of demolished concrete structures and pavements as aggregate in new concrete constructions. Because of the increasing waste production and growing public awareness of environmental problems, recycled materials from demolished concrete or masonry can be profitably used in a variety of ways in the construction industry.

It has been estimated that, approximately, 50 million tons of concrete are demolished each year in the European Economic Communities,[60] 60 million tons in the USA,[61,62] 10–12 million tons in Japan,[63] 13 million tons in France,[64] 13.5 million tons in Hong Kong,[65] and over 3.0 million tons in Australia.[66]

Due to the rapid development in research on the use of recycled concrete aggregate for the production of new concrete, the production of high-strength and high-performance concrete is now possible.[67] The use of the coarse recycled concrete aggregate up to 30% has been recommended, but the addition of super plasticisers is often considered necessary for achieving the required workability of new concrete.[68,69]

Three types of recycled aggregate are specified by RILEM.[70]

1. Type I, which consists primarily of masonry rubble,
2. Type II, which consists primarily of concrete rubble, and
3. Type III, which consists of a blend of recycled aggregate (max. 20%) and natural aggregates (min. 80%).

Mulheron[71] distinguishes the following four main categories of construction and demolition waste:

1. Clean crushed concrete which consists crushed and graded concrete containing less than 5% of brick or other stony material.
2. Clean crushed brick which consists crushed and graded brick containing less than 5% of other material such as concrete or natural stone.
3. Clean demolition debris which consist of crushed and graded concrete and brick.
4. Crushed demolition debris which consist of mixed and crushed concrete, and brick that has been screened and stored to remove excessive contamination, but still containing a proportion of wood, glass, or other impurities.

Any crushed aggregate produced from construction and demolition waste tends to be referred to as crushed concrete,[72] but for road construction, the term has normally referred to crushed concrete produced from the break-up and crushing of concrete slabs from road and airfield pavements.

According to a Dutch classification[73] for recycled granular base materials, crushed concrete is defined to contain:

- Main components:
    1. At least 80% (by weight) of crushed gravel or crushed concrete aggregate.
    2. At most 10% (by weight) of other broken stony material, the particles of which have a particle density of at least 2.1 Mg/m^3 (2.1 g/cm^3).

- Additional elements:
    1. At most 10% (by weight) of other crushed stone or stony material, or at most 5% asphalt.
- Impurities:
    1. At most 1% nonstony material (plaster, plastic, rubber, etc.).
    2. At most 0.1% decomposable organic matter such as wood and vegetable remains.

4.1. *Production of Recycled Aggregate*

Plants for the processing of recycled aggregates are almost similar to the plants for the processing of natural aggregates. They include various types of crushers, screens, magnetic devices for iron and steel removal and other devices for the removal of undesired material and aggregate transfer equipment (Fig. 8).

Demolition of buildings, bridges, and other structures is carried out by various methods; controlled explosive demolition and the use of wrecking balls are the most common. The bigger chunks of plain or reinforced concrete are then broken into smaller sizes with vehicle-mounted impact hammers and stockpiled on the ground.

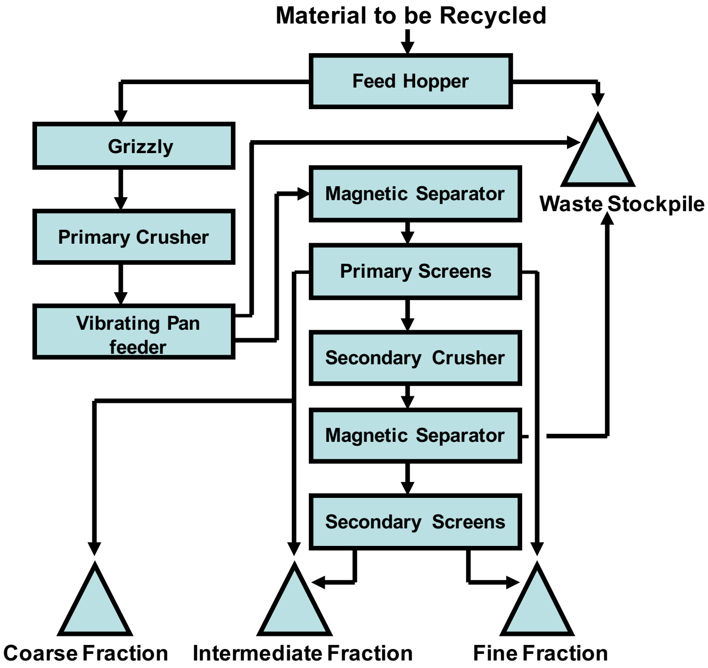

Figure 8. Flow Diagram of Aggregate Recycling Operation (*Source*: Ref. [85]).

Impact crushers, jaw crushers or hydraulic breakers are then used to reduce the demolished rubble to a suitable size for further handling.

When recycled concrete aggregate is collected from different sources or types of concrete, it should be placed in separate stockpiles or blended with other aggregates to ensure the uniformity of the properties of recycled aggregates. To produce aggregates within the limits of concrete mix gradation, various types of crushing and screening is required.

The aggregate processing plant may be either an open or closed system. The closed system is preferred because it allows greater control over the maximum aggregate size produced and produces the finished product to a higher uniformity.[69] The schematic diagrams of different types of recycled aggregate processing are illustrated in Figs. 9 and 10, both of these processes are regarded as "first generation plants" since they do not incorporate facilities to remove contaminants.

Second generation or closed system plants (as illustrated in Fig. 11) incorporate the facilities for the removal of contaminants and produce uniformly-sized particles of the end product.

4.1.1. *Crushers*

A study carried out in the Netherlands[74] on the feasibility of different types of crushers for the processing of recycled aggregate concluded that jaw crushers provide the particle size distribution of recycled aggregate required for concrete production.

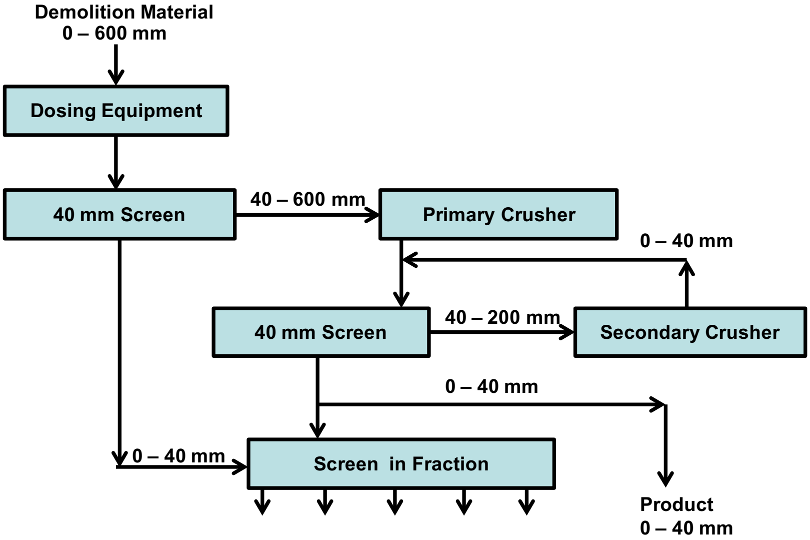

Figure 9. Flowchart of Typical Plant for Closed-System Production of Recycled Aggregate from Concrete Debris that is Free from Foreign Matter (Cited by Ref. [67]).

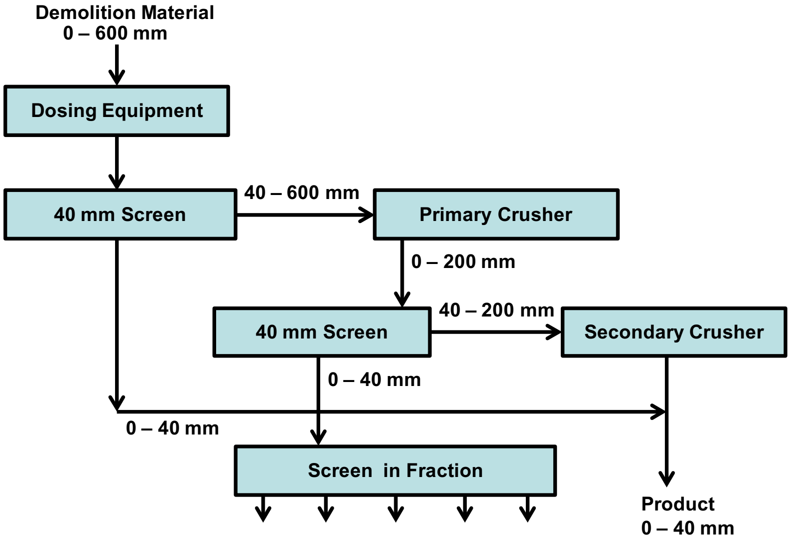

Figure 10. Flowchart of Typical Plant for Open-System Production of Recycled Aggregate from Concrete Debris that is Free from Foreign Matter (Cited by Ref. [67]).

Impact crushers were found to provide better particle size distribution for road construction, and were less sensitive to materials that cannot be crushed such as reinforcing bars. Cone crusher can be effectively used to process concrete with maximum feed size of 200 mm. Swing hammer mills are seldom used for recycled aggregate processing. The disadvantages of using impact crushers are, that they crush not only old mortar but aggregate also, thus producing a low-quality aggregate. They are also higher on maintenance costs because of high wear and tear.

To recycle demolition concrete waste, effective methods of demolition should be used to reduce the maximum aggregate size to approximately 1200 mm for fixed plants and 400–700 mm for mobile plants.

4.1.2. *Processing*

The recycled concrete aggregates are required to be free from all contaminants such as dirt, clay lumps, gypsum plaster, wood, plastics, paper, paint, asphalt and other impurities. Most lightweight materials can be removed from crushed debris and brought to the level of acceptance as per specifications by wet sieving.[75] Heimsoth[76] claimed that the same could be achieved by dry screening process when impurities are heavier than water. Lightweight and fine particulate impurities can be removed from rubble by air classification processes. Various methods available for the sorting of crushed debris have been reviewed elaborately by Drees.[77]

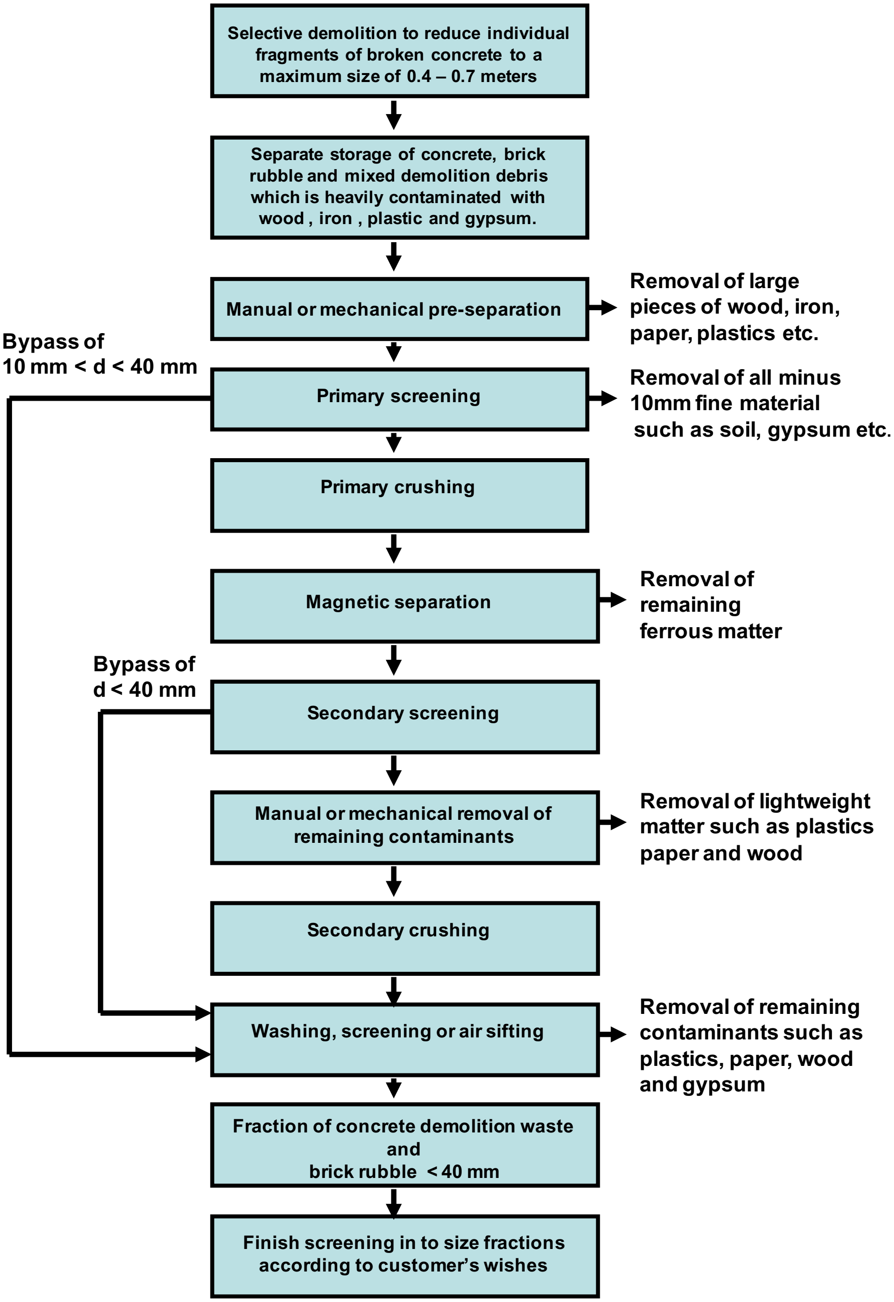

Figure 11. Processing Procedure for Building and Demolition Waste (Cited by Ref. [67]).

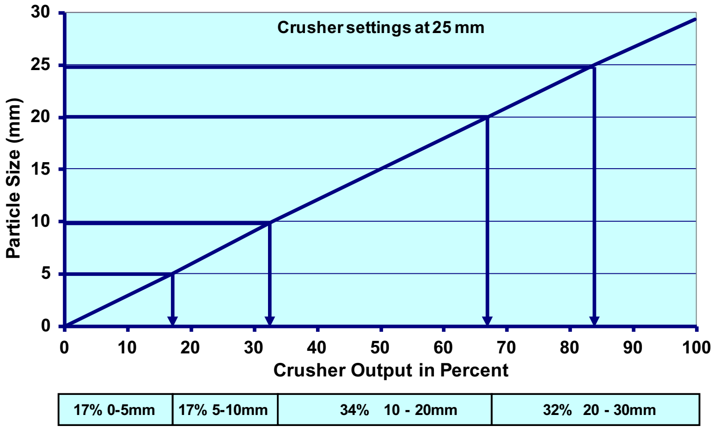

Figure 12. Correlation Between Crusher Setting and Particle Size Distribution of Crusher Products (Cited by Ref. [67]).

Table 8. Typical Average Sieve Analysis of 14-mm Recycled Aggregate.

Coarse Aggregate	Percentage of Weight Passing Through Sieve						
Max. size (mm)	19	13.5	9.5	6.7	4.75	2.3	150 μm
Recycled aggregate	100	91.4	28.7	7.6	5.4	4.2	0.5 μm

4.1.3. *Grading of Crusher Products*

It is assumed that the material sizes produced in crushers have a linear particle size distribution, as illustrated in Fig. 12. Plant grading of recycled concrete aggregate depends primarily on the crushing process and setting of sieve sizes to achieve the recommended nominal uniform grading.

Some variation in sieve analysis of coarse recycled aggregate can occur for material obtained from different sources. Thus, the blending of the mixture of different sizes should be uniform. Table 8 shows the typical grading of aggregates for recycled aggregates.

4.2. *Impurities*

To ensure that the finished concrete has consistent strength and durability, impurities such as sulfate and chloride ions and alkali reactive aggregate that can cause breakdown of concrete causing D-cracking in concrete pavements must be controlled.[78,79] D-cracks are closely spaced cracks parallel to transverse and longitudinal joints protruding outward to the center of the pavement panel. These cracks typically start in the saturated aggregate at the base of the pavement and progress upward. It has been recommended that the degree of contamination and potential reactivity of recycled aggregate should remain in the limits for natural aggregates.[80]

4.3. *Material Properties*

4.3.1. *Physical Properties*

To evaluate the utilization of recycled concrete aggregate, some of the engineering properties that are of particular interest include gradation, particle shape, specific gravity, water absorption, moisture content, aggregate strength, permeability, degree of contamination and durability.

4.3.1.1. Gradation

Recycled concrete aggregate should meet the AASHTO M 6-08[81] and MP 16-10[82] gradation requirements for hydraulic cement concrete. A plant can produce any desired gradation with some appropriate adjustments. Crushed fine aggregates (No. 4 sieve, minus 4.75 mm) are generally not used or blended with natural sand.

4.3.1.2. Shape

Processed recycled aggregate, being 100% crushed material, possess a highly angular shape. While this shape assists in increasing the strength of the mix, it can reduce its workability.

4.3.1.3. Specific gravity

The specific gravity of processed recycled aggregate ranges from 2.0 to 2.5, which is generally lower than natural aggregate mainly because it is relatively porous, and also due to the adhesion of less dense residual mortar lumps or particles to the surface of the original natural aggregates. The difference becomes more pronounced with decreasing particle size. The volume percentage of residual mortar increases with increasing particle size; correspondingly, the water absorption of recycled aggregate is increased. The specific gravity of processed recycled aggregate fines range from 2.0 to 2.3.[80] The specific gravity of the recycled aggregate constitutes a very important parameter for accurate batching and concrete mix design. The accuracy of batching is, in turn, affected by the moisture content of aggregate particles, which is partly influenced by variations in the composition of the recycled material.

4.3.1.4. Water absorption

Water absorption is one of the key performance indicators of recycled concrete aggregate (RCA) that define the quality of aggregate. The water absorption capacity of RCA is generally higher than the standard threshold for natural aggregates due to higher water absorption of old cement mortar attached to the aggregate particles.[66] Absorption values of coarse recycled aggregate vary from 3.7 to 8.74, while for fine recycled aggregate, values range from 8.3 to 12.1% as compared to about 2.0% for fine natural aggregate.[63,83] Water absorption of coarse and fine recycled aggregate can be determined according to BS-812 or ASTM C-127 and ASTM C-128, respectively.

Higher absorption rates pose potential problems for concrete production and can adversely affect concrete workability. A maximum of 10% water absorption is accepted for many construction applications. Pre-wetting of aggregate normally offsets inadequate concrete workability due to possible variations in aggregate absorption rates. The relation between water absorption and density of recycled aggregate was found to be parabolic by Kreijger.[84]

4.3.1.5. Moisture content

The moisture content of the *in-situ* stockpile of processed recycled aggregate is typically the same as that for conventional granular material.

4.3.1.6. Los Angles abrasion

The aggregate abrasion value is defined as the percentage loss in weight by abrasion, so that a low value indicates a high resistance to abrasion. Several tests are performed to evaluate the aggregate abrasion value, of these the Los Angeles abrasion test is more commonly used all over the world. Coarse grained RCA exhibit good abrasion resistance. ASTM C-33 requires that aggregates may be used for concrete production when the Los Angeles abrasion loss does not exceed 50%. Crushed stone for road construction purposes is normally required to have L.A abrasion loss not exceeding 40%.

Based on the data available, Hansen[67] concluded that RCA produced from all but the poorest quality concrete can be expected to meet the ASTM requirements to L.A abrasion loss percentage. Some typical physical properties of RCA are listed in Table 9. The Los Angeles abrasion value of RCA can be improved just by changing the crushing procedure. RCA crushed by an impact crusher showed lower Los Angeles abrasion values as compared to those processed by a jaw crusher which gave slightly higher values.

4.3.2. *Chemical Properties*

4.3.2.1. Deleterious substances

Deleterious substances, which can have potentially adverse effects on the properties of concrete, include chlorides, sulfates, alkali-reactive materials and components such as wood and plastics; these can influence the setting and durability of concrete.

The extent to which chemical contaminants are likely to adversely affect concrete properties can be analyzed using suitable leaching or related test methods.

Table 9. Typical Physical Properties of Processed Recycled Concrete Aggregate.

Property	Value
Specific gravity	
■ Coarse particles	2.2–2.5
■ Fine particles	2.0–2.3
Absorption percent	
■ Coarse particles	2.0–6.0
■ Fine particles	4.0–8.0[a]
Los Angeles abrasion loss percentage (ASTM C131)	
■ Coarse particles	20–45[b]

[a] Absorption values as high as 11.8% have been reported.[82]

[b] *Source*: Refs. [80, 82, 84].

4.3.2.2. Chlorides

Chloride contamination of RCA that are derived from marine structures or exposed structural elements, and that which are the result of many years of deicing salt application on pavements can induce the corrosion of steel reinforcements embedded in the new concrete. Depending on the source of recycled aggregate, chloride may be present in the form of magnesium or sodium chloride. Standards of many countries have put a limit on the total chloride content for various construction applications; for reinforced concrete, 0.05% is allowed, while for non structural concrete the percentage is 1.0% and for pre-stressed concrete, 0.015%.[83,122]

4.3.2.3. Sulfates

In recycled concrete aggregates, sulfates can be present as gypsum and as sulfates from admixtures or binders. In mixed aggregates and recycled concrete aggregate (RCA) in general, their presence is attributed to residues of plasterwork, insulation prefabricated pieces, drywall, etc. While the presence of gypsum is potentially harmful to concrete due to the deleterious expansive reactions within the cement paste resulting in the formation of ettringite and thaumausite, also, from an environmental point of view, it is important to prevent the excessive leaching of the sulfates. The need to limit the presence of sulfates in recycled aggregate is a common necessity. The environment surrounding the material, is of particular importance as it varies from place to place — affecting the overall product and leaching release. The sulfate contamination of final products is of particular concern in countries with wide use of gypsum in construction, such as Germany and Spain.

For sulfate soundness, the RCA may be tested by ASTM C-33, which specifies the limits in loss in weight when aggregate is subjected to five cycles of alternative soaking and drying in a sulfate solution. The test is carried out according to ASTM

C-88. The weight loss of coarse and fine aggregate is limited by ASTM C-33 to 12 and 10% when sodium sulfate is used, and 18 and 15% when magnesium sulfate is used, respectively. Investigations by Kobayashi *et al.*[85] reveal a higher percentage of weight loss in RCA ranging from 22.7% to 31.5%.

An effective technique for reducing gypsum content is selection by means of near-infrared spectroscopy. This newly developed technique has been successful in many processing plants around the world.

4.3.2.4. Alkali-silica reactive aggregates

Recycled concrete aggregates may also contain material susceptible to alkali-silica reaction, such as reactive quartz and plate glass, leading to potentially damaging expansive reactions under wet conditions. Three things are necessary to cause dangerous alkali aggregate reactions in concrete:

1. An aggregate with sufficient amounts of reactive constituents that are soluble in highly alkaline aqueous solutions.
2. Enough water-soluble alkali from cement to increase the pore-liquid's pH up to 14 and hold it there, initiating the production of alkali silica gel.
3. Sufficient water to maintain the solutions and provide moisture for the swelling of the gel.

The risk of damage due to alkali silica reactivity can be minimized by meeting the following set of conditions:[86]

1. Na_2O equivalent content must not be more than 0.2 kg/100 kg of recycled concrete aggregate (RCA).
2. The total calculated content of alkali of all concrete constituents (OPC Type I + coarse aggregate + fine aggregate + fly ash + admixture + water added) does not exceed: 3.5 kg/m3 Na_2O equivalent where the declared average content of alkali in OPC Type I component (of a combination) does not exceed 0.75%; or 3.0 kg/m^3 Na_2O equivalent where the declared average content of alkali in OPC Type I component (of a combination) is 0.76% or higher.
3. The amounts of alkali in blast furnace slag and fly ash does not exceed 1% and 5% Na_2O equivalent respectively.
4. Aggregates other than recycled concrete aggregate are not classified as highly reactive.

4.3.2.5. Fine material

The reclaimed concrete aggregates should not contain more than 1.5% of dust or material finer than 75 μm (No. 200 sieve) nor clay lumps, chert, friable particles, coal and lignite, or other deleterious substances that exceed the maximum allowable amount listed in Table 10.

Table 10. Limits of Deleterious Substances and Physical Property Requirements of Coarse Aggregate for Concrete. *Source* Ref. [82].

	Maximum Allowable Percentage[a]				
Class Designation[b]	Clay Lumps and Friable Particles	Chert (Less than 2.40 sp. gr. SSD)[c]	Sum of Clay Lumps, Friable Particles, and Chert (Less than 2040 sp. gr. SSD)[c]	Other Deleterious Substances[d]	Coal and Lignite
A	2.0	3.0	2.0	0.3	0.2
B	3.0	5.0	3.0	0.3	0.2
C	3.0	8.0	5.0	0.3	0.2

Notes:

[a] The engineer may supplement the requirement of Table 10 by placing limits on the amount of deleterious substances or physical properties in accordance with local experience and practice.

[b] Aggregate conforming to the requirements for the various classes designated in Table 10 should generally be suitable for the following uses:

Typical Uses (Suggested)	**Weathering Exposure**	**Class of Aggregate**
Concrete pavements, cement treated base courses, sidewalks, median barriers, curbing and other non-structural applications where a moderate number of pop outs can be tolerated.	Severe	A
	Moderate	B
	Negligible	C

The purchaser or specifier must indicate the class of aggregate to be furnished, as the degree of weathering exposure is not precisely defined.

[c] These limitations apply only to aggregates in which chert appears as an impurity. They are not applicable to gravels that are predominantly chert. Limitations on soundness of such aggregates must be based on service records in the environment in which they are used.

[d] Other deleterious substances include adherent fines, vegetable matter, plastics, plaster, paper, gypsum board, metals, fabrics, wood, brick, tile, glass, and bituminous materials. The percentage of these materials shall be determined in accordance with ASTM C 295 or other equivalent method approved by the specifying jurisdiction.

4.3.2.6. Organic substances

Organic substances such as polymeric materials, wood, paper, textile fabric and joint seals may become unstable in concrete when subjected to cyclic wet and dry or freeze thaw conditions. Hence, a limit of 0.15% of organic substance by weight of aggregate (about 2 kg/m^3) for substances lighter than 1200 kg/m^3 is generally recommended.[88]

Owing to their lower densities, the content of organic impurities increase in concrete in terms of parts per volume. Other organic substances such as paint may entrain considerable amounts of air in concrete.

4.3.3. *Mechanical Properties*

4.3.3.1. Compressive strength and rate of strength development

On the basis of the research conducted by Refs. 87–91, it can safely be concluded that the compressive strength of recycled aggregate concrete is 10–25% lower than the compressive strength of conventional concrete. The compressive strength of the recycled concrete was found to be dependent on the strength of the original concrete, and largely controlled by a combined effect of the water–cement ratio of the original concrete and the water–cement ratio of the recycled concrete, when all the other variables were kept identical.[92,93] In addition, no increase was experienced in the compressive strength of recycled aggregate concrete when the water–cement ratio was decreased below 0.4. Hansen *et al.*[94] also found that it was the use of fine recycled aggregate that reduced the compressive strength of recycled concrete as compared to coarse recycled aggregate. Considering the fact that fine recycled aggregates also increase the water requirement of recycled concrete and that the frost resistance of such concretes is doubtful, it has been suggested that recycled fine aggregate should not be used for the production of quality concrete.

In a recent study,[95] the authors conclude that the concrete made with recycled aggregates (20, 50, and 100% replacements) from recycled concrete and blended with 35% blast furnace slag cement can have the same workability and can achieve the same compressive strength at 28 days as compared to the concretes made from natural aggregates in the range of 20–40 MPa.

4.3.3.2. Drying shrinkage

All concrete and mortar shrink on drying out due to evaporation and loss of unconsumed water (*i.e.* that does not take part in the cement hydration). The shrinkage of concrete can affect several mechanical and other durability properties of concrete due to the formation of micro-cracks. Normally the incorporation of recycled concrete

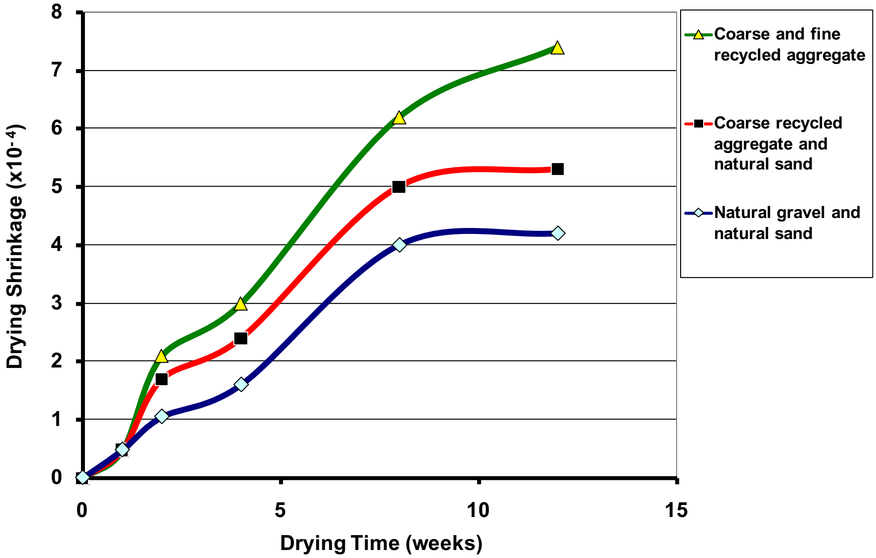

Figure 13. Drying Shrinkage of Original and Recycled Aggregate Concretes as Function of Time (*Source*: Ref. [96]).

aggregate in concrete increases the drying shrinkage mainly due to the increase in paste content in concrete.

Hasaba *et al.*[96] found that the drying shrinkage of recycled aggregate concrete made with cement content of 300 kg/m^3, coarse recycled aggregate and fine recycled aggregate, to be 70% larger than the drying shrinkage of the corresponding control concrete made with natural aggregates. The drying shrinkage was found to be 50% larger than that of the control when coarse recycled aggregate and natural sand were used (Fig. 13).

Domingo-Cabo *et al.*[97] measured drying shrinkage of natural and recycled concrete aggregate (RCA) with replacement of 20, 50 and 100% recycled concrete aggregate (RCA). After 180 days, they observed 20% and 70% higher shrinkage for recycled aggregate concrete (RAC) made with 50 and 100% incorporation of RCA, respectively. The reason for higher shrinkage was attributed to the increase in volume of the cement paste and the porosity of concrete due to the incorporation of RCA.

Limbachiya and Meddah *et al.*[98] reported a reduction in drying shrinkage of up to 37% in conventional concrete when 30% of OPC was replaced with fly ash and various water/cement ratios, due to the reduction in water demand in concrete mix because of the lubricating effect of fly ash and their high capacity of water retention which contributes to minimize the water available in the pore network for external drying. Regardless of the type of binder used, the shrinkage of concrete gradually increased with the increase in RCA content due to higher water absorption capacity and adhered mortar content on RCA. It was also noted that no significant increase of drying shrinkage was observed when 30% RCA was incorporated in both types of concrete. The lowest shrinkage was observed with control fly ash concrete while the largest shrinkage was obtained for RAC made with 100% replacement with RCA.

Kou and Poon *et al.*[91] studied the performance of natural and recycled concrete aggregate (RCA) prepared with the incorporation of silica fume, fly ash, metakaolin and ground blast furnace slag. They reported that the drying shrinkage of concrete containing silica fume and metakaolin as 10 and 15% replacement of OPC was higher than that of conventional concrete due to the pozzolanic reaction and formation of higher amount of calcium silicate hydrate gel. On the other hand, concrete mixtures containing fly ash and ground blast furnace slag at 35 and 55% replacement showed lower drying shrinkage than that of conventional concrete due to lower hydration rate of fly ash and ground blast furnace slag as well as the restraining effect of the unhydrated powder particles in cement paste.

4.3.3.3. Modulus of elasticity

The modulus of elasticity of recycled aggregate concrete (RAC) remains lower than that of conventional concrete made with natural aggregate because concrete's elastic modulus is intrinsically associated to aggregate's volumetric fraction, specific gravity, modulus of elasticity of aggregate as well as cement matrix and characteristics of transition zone. Aggregate's deformation is associated mainly to its porosity and to a lesser degree to maximum dimension of aggregate, its form, texture, grading and mineralogical composition. It is the aggregate's modulus that controls the restriction capacity of the matrix deformation and this is controlled by the porosity of the aggregate.

Ravindrarajah, Loo, and Tam[99] found that the modulus of elasticity of RAC was lower than that of natural aggregate concrete, and that the difference increased with strength. The reductions in static and dynamic moduli of elasticity, for a medium strength concrete, was about 25% for water-cured concrete and 35% for air-cured concrete. Recycled fine aggregate reduced the modulus of elasticity by a similar amount as RCA.

Shi Cong Kou *et al.*[100] in their study used two water/binder ratios of 0.45 and 0.55 with RCA replacement of 0, 20, 50 and 100% by weight of natural aggregate. In addition, fly ash was used as 0, 25 and 35% replacement of cement in the concrete mix. He concluded that the static moduluus of elasticity decreased as the RCA content increased. However, the reduction could be adequately compensated by using lower water/binder ratio. At the same RCA replacement level and water/binder ratio, the use of fly ash as partial replacement of cement decreased the static modulus of elasticity.

Domingo-Cabo *et al.*[97] found a decrease of the 28-day modulus of elasticity due to the incorporation of good quality RCA as 0, 20, 50 and 100% replacement of coarse natural aggregate in mixes with similar water/cement ratio. Unlike most investigations, the RCA used in this study was not pre-saturated before concrete mixing — instead a superplasticizer was used to prepare a workable mix. After the RAC was prepared at constant slump by considering the amount of water absorbed by RCA, the 28-day modulus of elasticity was similar to that of conventional concrete.

Wang *et al.*[101] by using sea water and dry-wet circulation to simulate marine environment, tested the RCA with different replacement percentages of 0, 30 and 60%, and immersed in sea water for 4, 8, 12 and 16 months, observed that (1) the modulus of elasticity decreased with RCA replacement and (2) the marine environment also had considerable influence on modulus of elasticity. It was lower than that of conventional concrete. It decreased with increasing RCA replacement percentage as well the corroding time. When the RCA replacement was 60%, the modulus of elasticity decreased by 11.0%, and it decreased by 14.0% at 8 months immersion time while it decreased by 23.5% when the immersion time in seawater was increased to 16 months. They interpret that the destruction process accelerates after internal bonding is gradually destroyed.

4.4. *Potential Outlets of Recycled Concrete Aggregates*

Recycling of crushed concrete into new construction applications not only offers a way to reduce waste disposal loads sent to area landfills but also extends the life of natural resources by supplementing resource supply. Approximately 25–40% of construction and demolition waste consists of masonry rubble, concrete, mortar, and brick that are potentially recycleable as aggregate.[102] It has been estimated that in the USA, the production rate of recycled aggregate is less than 5% of annual production of natural aggregate.[103] The supply of crushed concrete rubble generated by construction activities each year is estimated to be 100 million tons.[104]

In Europe, there has been a large progress in the processing and recycling of demolition waste, since the start in the Netherlands at the beginning of the 1980s. The major driving forces for many countries (e.g. the Netherlands, Germany, and Denmark) have been and still are the limited landfill capacity and reduction of adequate mineral resources.[105,106] The level of recycling of demolition waste in Norway, in contrast is quite low, roughly estimated to be 10–20%, which is well below the European Union average of 25%.[107] The outlook of the goals setup by the Organisation for Economic Co-operation and Development (OECD) countries[108] is illustrated in Table 11.

Crushed concrete is regarded as a replacement for natural aggregates for virtually all uses, which include highways, roads, and buildings. Figure 14 illustrates

Table 11. Goals for Waste Reduction in OECD Countries.

Country	Goals
Australia	Reduce waste going to landfills by 50% by the year 2000
Austria	Long-term target of 90% reuse of road materials
Belgium	To recycle 70% of all building and construction waste and to reduce landfill to 30%
Denmark	Reuse 54% of the total amount of waste in the year 2000 Reuse 60% of construction and demolition waste Reuse 65% of slag from incineration
France	Reduce traditional landfill to zero in the year 2002
Japan	Reuse generation of waste by 10% and increase reuse from 42% to 80% by the year 2000
The Netherlands	Reuse 90% of construction and demolition material in 2000 and continue 100% reuse of MSWI bottom ash
United Kingdom	Double the use of recycled materials with in 15 years

Source: Ref. [108].

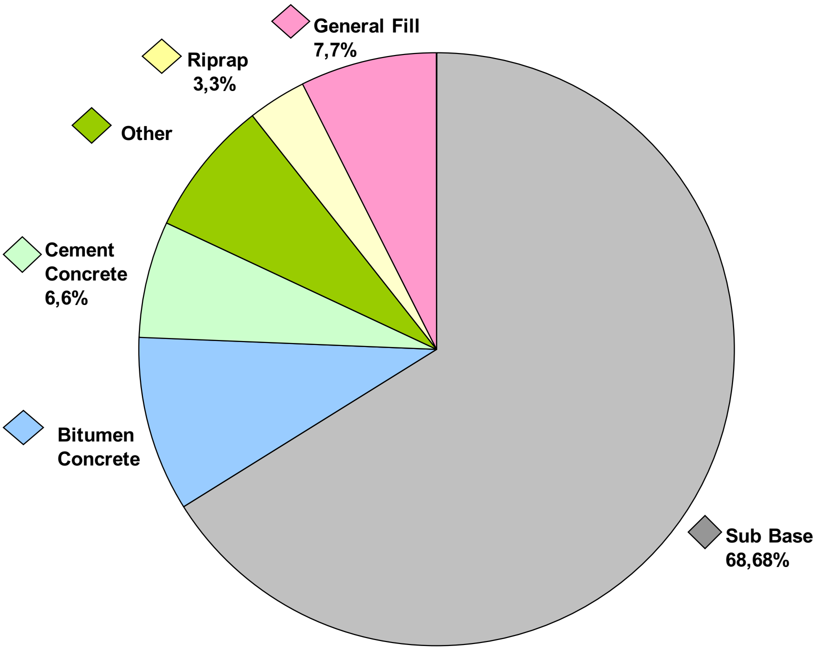

Figure 14. End Uses of Recycled Concrete Aggregate (*Source*: Ref. [109]).

these end uses based on their percentage consumption. All the uses are substitutions for natural aggregates derived from crushed stone, gravel, and sand. Sub-base is the aggregate material used for the bottom 30–60 cm in highway construction. Bituminous concrete is the pavement layer for highways that use approximately 4.5–6% by weight asphalt binder mixed with 94–95.5% aggregate material for the road surface. Cement concrete is the combination of Portland cement, water and 60–75% aggregate used for highway and building constructions. General fill is aggregate used to provide drainage for building foundations, pipe beddings, or leach fields. Riprap is large chunks of aggregate used for shore protection along rivers, harbors and the areas where erosion damage is possible. The "other" category includes end uses such as railroad ballast and landscape rock.

4.4.1. *Road Construction*

4.4.1.1. Bulk fill

If the required level of a new road is not the same as that of the ground over which it has to be built and has to be lowered or raised, the processes required are known as "cut" and "fill." The fill required may be in large quantities and, therefore, should be easy to transport, to place, and to compact. After compaction, it shall provide a stable bed, strong enough to receive the layer above it, which may be the capping layer or the sub-base, depending on the circumstance. Although crushed concrete and crushed brick can be used as bulk fill, this use is wasteful because they are suitable for use in more demanding situations.

4.4.1.2. Selective fill

Selective fills have to meet stringent requirements as compared to bulk fill, since they are used in situations such as bedding material for drains and as the fill placed in the immediate vicinity of bridges and structures. Crushed concrete can be used for most applications. Limits for use, which are not stringent, are specified in terms of grading, plasticity, particle strength, and chemical properties.

4.4.1.3. Capping layer

On weak sub grades, it is common practice to use a capping layer between the subgrade and the sub-base. This reduces the thickness of subbase and provides a suitably firm surface for the placement and compaction of the subbase. The capping layer can be constructed from low-cost granular material to a lower specification than the subbase.

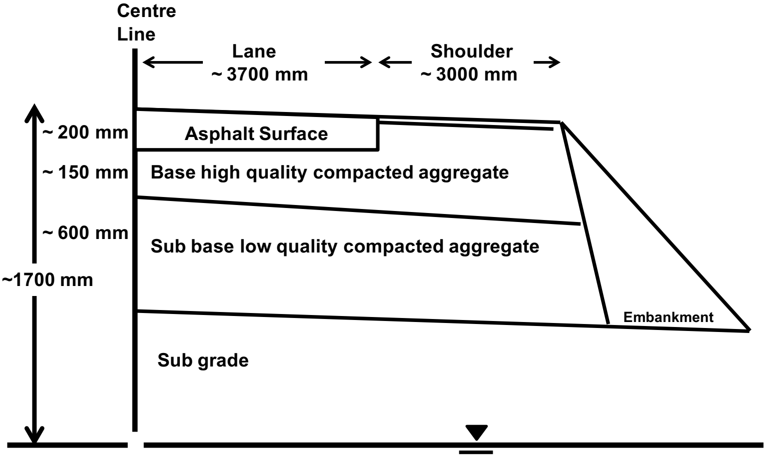

Figure 15. Schematic Diagram of a Typical Road Construction Cross section (Approximate Dimensions) (*Source*: Ref. [110]).

4.4.1.4. Sub base and base layers

Roads are typically constructed in layers of compacted materials generally increasing in quality from subgrade to road surface (Fig. 15). The surface layer is usually asphalt (bitumen binder and aggregate) though previously road surfaces were also made with concrete. The subbase and base layers are made from compacted aggregate. The material used in each layer has to conform to specifications, which include grading, particle strength, and resistance to frost.

Because of the intensity of loading to be experienced by the pavement surface, the recycled concrete aggregates are preferably used in the lower layers of pavement. In this role, the absolute strength is less important but the consumption will be high due to the volume of material required in these layers.

4.4.2. *Concrete Pavements*

The specifications for Texas, Minnesota, and Michigan allow the use of recycled concrete aggregate in concrete pavements. Due to the dwindling supply of natural aggregates and the knowledge that their best aggregates have been used in their oldest pavements, Texas is using recycled concrete aggregates as a viable aggregate in Dallas and Houston metro areas. The Texas Department of Transportation (Tx DOT) has found that the use of recycled concrete aggregates in concrete pavements gave a satisfactory performance, allowing them to design new pavements with recycled concrete aggregates as aggregate. In their specifications, San Francisco allows the use of recycled concrete aggregates in all nonstructural flat works, which include sidewalks, curbs and other features other than pavements.

4.4.3. *Building Construction*

4.4.3.1. Hardcore

Recycled concrete aggregate is also utilized as hardcore filling in building construction provided that the grading of the recycled aggregate is checked to suit the condition of the soil.

4.4.3.2. Moulded concrete bricks and blocks

A study carried out by Poon, Kou, and Lam[111] concluded that the replacement of coarse and fine natural aggregates with recycled aggregates at the levels of 25% and 50% had little effect on the compressive strength of the block and brick specimen, but higher levels of replacement reduced the compressive strength. Using recycled aggregates at the level of up to 100%, concrete paving blocks with 28 days compressive strength of not less than 49 Mpa can be produced. The performance of the blocks and bricks was also found satisfactory for shrinkage and skid resistance tests.

4.4.4. *Oyster Beds*

Virginia State in the USA has found a novel way for the utilization of recycled aggregates. An artificial reef is created using recycled concrete aggregates, which is then covered with crushed oyster shells, creating an oyster bed. Since the material is being replaced in a saline environment, concrete with high chloride content is acceptable.

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Chapter 11

USE OF SOLID WASTES AS CONSTRUCTION MATERIALS

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Abstract

The use of solid wastes as construction materials is one way to address the global issue of sustainability and climate change. Many researchers and scientists have made significant contribution to turn wastes into useful resources. This chapter summarizes the publications of researchers who spent years of intensive effort to investigate certain important aspects of solid wastes utilization in construction. The solid wastes include, fly ash, timber industrial ash (TIA), palm oil fuel ash, expanded coal slag (ECS), foamed aggregate, waste styrofoam, and ground granulated blast furnace slag. The chemical composition, physical and engineering properties of the solid wastes and their effect on the performance of concrete, the most widely used construction material is briefly covered. Sustainable and viable options for infrastructure development are explored and discussed. The controlled density pervious lightweight concrete with high porosity allows water from precipitation and other sources to pass through it, thereby reducing the surface runoff from going straight into the drain. The recharging of water into designated retention ponds are designed to minimize the fluctuation of groundwater levels. The study is to establish an economical method for construction on problematic soils especially on peat. The pervious low-carbon media with TIA has been studied on its potential for harvested rainwater treatment. The compressive strength development and water absorption of ECS concrete were presented in the ECS mix design nomograph that can be used to predict the dry density, compressive strength, and water absorption of various percentages of ECS concrete. The structural performance of reinforced concrete beams containing recycled crushed foamed concrete fine aggregate (RFA) is briefly covered. The cement industry contributes about 5% to global CO_2 emissions from the calcination process of limestone and combustion of fuels in the kiln. A model of CO_2 uptake by biomass silica foamed concrete is proposed as a potential mitigation strategy against CO_2 emission. The proposed foamed concrete absorbs CO_2 42.7% faster than the normal concrete. Successful deployment could contribute toward sustainable development while benefiting from the carbon credits.

Keywords: Light weight concrete, timber industrial ash, biomass aggregate, micronized silica, fly ash, biomass silica foamed concrete.

1. Introduction

This chapter summarizes the research achievements of selected researchers and reports the experimental findings of a long-term intensive research program focused on the exploitation of solid wastes in construction conducted by the authors. The objective of the research program is to develop an environmental-friendly and economical process for the exploitation of solid wastes in construction. The solid wastes experimented include timber industrial ash (TIA), rice husk ash, fly ash, and palm oil fuel ash. Other solid wastes include biomass particularly rice husk, palm fiber, paper sludge and palm clinker. Packaging wastes such as recycled expanded polystyrene (EPS) are also explored.

An engineered foamed grout and shear wall system with the combination of recycled EPS and lightweight concrete has been developed. It provides an environment-friendly and economical solution to construction on soft soil, toward providing affordable quality-assured fast-track method and to enhance the competitive edge of the construction industry. Obstacles toward achieving commercialization are briefly discussed.

1.1. *Exploiting Wastes in Concrete*

Among researchers who have made significant contribution to knowledge include Samarin[1] who reviewed the impact of Kyoto Protokol. New levels of abatement of greenhouse gas emissions by the year 2012 was stipulated, on the utilization of cement extenders in concrete, and on the use of wastes as new materials. Naik *et al.*[2] studied the use of industrial by-products in cement-based materials. He quoted that nearly 4.2 billion tonnes of nonhazardous by-products are generated from agricultural, domestic, industrial, and mineral sources. He emphasizes the need for recycling to conserve natural resources and to provide technical and economic benefits. Shoya *et al.*[3] studied the properties of self-compacting concrete with slag fine aggregates. He conducted the slump-flow test, the V-type funnel test, and filling vessel tests for the measurement of self-compatibility of fresh concrete. Mellmann *et al.*[4] investigated the exploitation of processed concrete rubble for reuse as aggregates. He described the feasibility of producing high-grade concrete from the rubble. Rad and Bonner[5] studied the properties and performance of recycled cementitious mortars. They attempted to link the petrological observations to the microstructural characteristics of cement gel interaction and activation. Bijen[6] proposed the use of secondary materials as a contribution to sustainability. He assessed the environmental impact in accordance with the draft ISO standards 14040 series. Ledham *et al.*[7] experimented on simple treatments to reduce the sensitivity to water of clayey concretes lightened by wood aggregates. They envisaged the potential use of local materials with hydraulic lime coating for the production of insulation materials. Mimoune *et al.*[8] used fly ash in the compound wood-cement. They concluded that the addition of fly ash did not give any amelioration and did not resolve elasticity problems during decompression.

2. Ash Utilization Research

2.1. *Fly Ash*

Coal ash production and utilization in the world with emphasis on its use in concrete was studied by Manz and Stewart.[9,10] They reviewed the worldwide production and utilization of coal ash from 1959 to 1989 for the UN Group of Experts on the Utilization of Ash. Quantities of fly ash recycled in each sector of the construction industry are given. According to Wang,[11] the fly-ash production in China was approximately 100 million tons per year in 1997. Puch and Vom Berg[12] reported that the coal combustion by-products in Germany are being recycled at a rate over 98%.

Dube,[13] Uchida,[14] Bland *et al.*,[15] and Zysk and Volke,[16] and Colmar[17] and others[18,19] review the use and utilization of fly ash in cement manufacturing and construction.

Jagiella[20] studied the utilization of coal combustion by-products such as fly ash, bottom ash, boiler slag, flue gas desulfurization (FGD) material, and fluidized bed combustion (FBC) by-products. Barringer[21] proposed a proportioning method based on the price, availability, and strength-gaining ability of the fly ash.

2.2. *Materials Characterization*

Galpern *et al.*[22] discuss the properties of ashes from fluidized bed with and without addition of lime/carbonate. They have reported that addition of up to 20–30% of low carbonate ash to cement does not affect the durability.

Ninomiya *et al.*[23] studied the effects of adding carbonate to ashes during melting. High temperature phase transformations were studied by x-ray diffraction (XRD) analysis and the glass phase was studied by FT-IR analysis. It was shown that addition of carbonates results in a glass phase with a higher pozzolanic reactivity than the parent coal glass.

Hower *et al.*[24] studied the quality of fly ash and its carbon content obtained from boilers converted to low-NO_x combustion. Other parameters include fineness and the relative amount of glass versus crystalline inorganic phases. It is shown that the post-conversion fly ash was higher in carbon than the pre-conversion ash from the same unit.

2.3. *Effect of Grinding*

Bouzoubaa *et al.*[25] studied the effect of grinding on the physical properties of ASTM Class F fly ashes. The specific gravity and the fineness are significantly increased with grinding up to two hours. Ruan *et al.*[26] used fluid energy mills to grind fly ash.

Paya *et al.*[27] demonstrated that the grinding fly ash increases its pozzolanic activity and enhanced the mechanical properties of mortar. A mathematical model was developed to predict the mechanical properties of mortars containing ground fly ash.

Kruger[28] described the means of separation of fly-ash particles to obtain fractions with different characteristics. These methods include air classification, electrostatic recovery, and density separation. Yoshida *et al.*[29] used an air-cyclone to classify particles of fly ash to a mean diameter 2 and 6 μm.

Murai *et al.*[30] reported that fly-ash particles of less than 10 μm affect the rheology, strength, water tightness, shrinkage, and durability of self-compacting concrete in a comparable or better way than ordinary fly ash, GGBS, and limestone filler.

Okoh *et al.*[31] investigated the kinetics of oxidation of residual carbon as a step toward producing low-carbon fly ash from high-carbon and low-quality fly ash.

2.4. *Lignite Fly Ash*

Meyrahn *et al.*[32] reported that lignite fly ashes have very low content of heavy metals and possess pozzolanic as well as hydraulic properties. Ludwig and Urbonas[33] and Bartscherer-Sauer *et al.*[34] studied granulometric and chemical-mineralogical properties of fly ash from lignite-fired power stations. Kahl *et al.*[35] compared the composition of lignite fly ashes produced in different power stations.

Dietz *et al.*[36] studied the composition, strength, and durability properties of German lignite fly ash. Volume stability, as well as carbonation depth, sulfate resistance, and freeze-thaw resistance were investigated. It was demonstrated that additions up to 10–15% do not lead to negative effects on fly-ash cement properties.

2.5. *Hydration and Microstructure*

Chang *et al.*[37] measured various aspects of workability, bleeding, setting, compressive and tensile strengths, pore distribution, and crystal structure of cement pastes containing fly ash from fluidized bed incineration systems.

Anthony *et al.*[38] studied the activation of unreacted calcium by means of injecting limestone into the furnace. They report hydrated tetracalcium aluminate as the major hydration product of the activation process.

Poon *et al.*[39] studied the influence of two different curing conditions (in water at 27°C, and in air at 15°C and 60% relative humidity) on the compressive strength, chloride, and water penetration properties of fly-ash cement pastes and mortars. Mercury intrusion porosimetry (MIP) showed that curing conditions has a significant influence on the strength, porosity, and durability of cement pastes and mortars.

Jalali *et al.*[40] derived a model based on nucleation and diffusional growth theories to predict the induction period of the reactions between lime and fly ash. Results indicate that accelerated curing can be used to estimate the induction period of any curing temperature.

Ma and Brown[41] studied the hydrothermal reactions of fly ash in the presence of calcium hydroxide or gypsum. Hydration rates were determined using isothermal calorimetry and XRD of hydrated phases. They further developed tobermorite from isothermal hydration of pure oxide mixtures that simulated fly ash.[42] Hassett and Eylands[43] proposed a calorimetric test procedure to measure the reactivity of fly ash based on the heat of hydration. Wang *et al.*[44] studied the kinetics of hydration reactions of fly ash and cement in the fly ash/cement mix.

2.6. *Effect of Carbon Particles*

Gao *et al.*[45] studied the specific sorptive behavior of carbon black that interacts strongly with the air-entraining admixtures. The adsorption of air-entraining admixtures is governed by the amount of accessible, hydrophobic, carbonaceous surface area while the degree of interaction was proportional to the nitrogen surface area.

Higaki *et al.*[46] devised an index based on an effective specific surface area of pores greater than 4 nm. Compared to other available techniques such as LOI, Blaine, and BET specific surface measurement, this index relates the air-entraining capacity of mortar to the suitability of sub-standard fly ash for the use in concrete.

Hwang *et al.*'s[47] study of several fly ashes indicated that a higher LOI value leads to a higher demand in superplasticizer to attain a given flow value. As the grain surface area increases, plastic viscosity and compressive strength increase.

2.7. *Strength*

Maltais and Marchand[48] conducted thermal analyses, compressive strength tests, and SEM observations on Class F fly ash. Results confirm that fly ash increases the rate of cement hydration at early ages. Increasing the curing temperature reduces the long-term compressive strength of the reference mortar mixture with no detrimental effect for the fly-ash mixtures.

Zhang *et al.*[49] studied concrete made with up to 50% and 60% weight content of high calcium-high SO_3 content fly ash. Although cement pastes containing fly ash showed volume instability and poor pore structure; however, mortars and concretes exhibited good performance with strength levels reaching 67 MPa after 90 days.

Wang and Hwang[50] studied the compressive strength and modulus of elasticity of fly-ash concrete at high temperatures (up to 440°C). Addition of fly ash counteracted the harm caused by high temperatures to cement pastes.

Kohno *et al.*[51] studied the effect of aggregate content on the strength of concrete with low- or high-quality fly ash. Sone *et al.*[52] proposed a mix design to replace the fine aggregates by fly ash. Kohno *et al.*[53] also used fly ash as a sand replacement.

Matsufuji and Koyama[54] developed a mix design method for high-volume fly-ash concrete. Ichikawa *et al.*[55] studied strength properties of high fly ash content concrete for different curing conditions. Pore distribution measurement indicated that even with air curing, fly-ash concrete has equivalent or better strength properties than the reference.

Krizan *et al.*[56] studied microstructural and strength properties of ground fly-ash cement, on pastes, mortars, and concrete using strength–porosity relationships. Porosity distribution was measured by acetone impregnation and squeezing specimens at high compressive stress levels followed by chemical analysis of pore solution and TGA-DTA.

3. Durability

3.1. *Sulfate Resistance*

Lizarraga[57] studied the behavior of C_3A-rich cement containing fly ash in presence of magnesium sulfate ($MgSO_4$) with satisfactory sulfate resistance results. Djuric *et al.*[58] and Miletic and Illic[59] compared the resistance of plain cements and those blended with fly ash to sulfate and ammonium ions solutions (C_3A content between 5% and 12%). Using flexural strength degradation as the durability criterion, the authors showed that fly-ash-blended cements were more resistant.

Illic *et al.*[60] studied sulfate resistance of Portland cement blended with lignite fly and bottom ash. The authors compared their experimental results to the durability prediction model of Matsufuji-Koyama. Krizan and Zivanovic[61] studied the sulfate resistance of fly-ash cements by the measurement of the flexural strength and ion-penetration by EDS.

3.2. *Chloride Ingress Resistance*

Sirivivatnanon and Kidav[62] reviewed the Australian and South-East Asian performance of fly-ash concrete with regards to chloride ingress. It is concluded that the higher the strength of the concrete is, the higher will be the chloride ingress resistance. Addition of slag or silica fume was beneficial to high-performance fly-ash concrete. Lee and Lee,[63] however, concluded that replacing 20% of cement by fly ash in a concrete containing calcium nitrite-based corrosion inhibitor, does not apparently improve the resistance to steel corrosion in sea water exposure.

Dhir *et al.*[64] studied the resistance of fly-ash concrete to chloride ingress. Chloride binding capacity, intrinsic permeability, and their concurrent influence on the coefficient of chloride diffusion. Chloride-binding capacity increased with chloride concentration and water/binder ratio. It was maximized at the 50% PFA replacement level. The coefficient of chloride diffusion was dependent on both the intrinsic permeability of the concrete and the ability of its cement matrix to bind chlorides.

Torii *et al.*[65] proposed a method using electric field to determine the diffusion coefficient of chlorides in concrete. Values of D_{Cl} in the range of 10^{-7} and 10^{-9} cm^2/s for cement–fly-ash pastes were obtained. The use of fly ash decreased the value of D_{Cl} after 28 days.

Wiens and Schiessl[66] proposed that the chloride binding effect of fly-ash cement is due to the increase in the sorptive surfaces of CSH and the irreversible binding of chloride by the hydrates. Using MIP, the refinement of the pore structure slowed the migration of chloride. A correlation was found between the migration coefficient and the threshold radius measured by MIP.

3.3. *Freeze-Thaw Resistance*

Marchand *et al.*[67] investigated the detrimental effect of adding fly ash to the resistance of concrete subjected to deicer salt scaling. Fly ash affected neither the freezable water content nor the total porosity indicating that the dense pore structure of fly-ash–cement systems is not responsible for the low resistance of fly-ash concrete to deicer salt scaling. Thomas[68] conducted a review of field and laboratory study of resistance of fly-ash concrete to salt scaling. It was demonstrated that mix designs that poorly performed in ASTM C672 accelerated scaling test, displayed an excellent resistance in the field. It was concluded that this test is not appropriate for fly-ash concrete.

Pavlenko *et al.*[69] studied the mix design of concrete containing slag and fly ash with a good resistance to freeze-thaw cycles. Kishida *et al.*[70] investigated the strength and durability properties of fly-ash concrete samples prepared by the immediate demolding method (compacted by strong vibration and compression). It was found that long-term strength of concrete with 10% addition of fly ash was comparable with that of concrete without fly ash. Freeze-thaw resistance decreased with the increase of fly ash, partly due to the reduction of entrained air. Fly-ash concrete displayed better performance with respect to chloride penetration and hydrochloric acid than concrete without fly ash.

3.4. *Other Durability Aspects*

Naik *et al.*[71] studied the influence of amount and origin of Class C fly ash on mechanical aspects and durability of concrete. It was observed that the origin of the fly ash has a strong influence on the early strength development while the drying shrinkage and abrasion resistance of fly-ash concrete was equivalent or higher than the control mixture. Abrasion resistance decreases, as fly-ash content increases.

Lee *et al.*[72] compared equivalent 28-days strength of fly-ash and OPC concretes. It was observed that the permeability decrease per fly-ash content increase is more pronounced in lower-strength concretes. Furthermore, the difference in permeability due to curing type was more important at lower strength levels and fly-ash contents.

Weinberg and Hemmings[73] tested landfill cells constructed using AFBC by-products and Class F coal fly ash over a five-year period. The reference cell, with no fly ash, rapidly deteriorated, whereas the cells with an FBC-fly-ash mix, showed improved strength and permeability over time. Due to the availability of soluble silica, the latter formed calcium silicate hydrate rather than thaumasite.

Keck and Riggs[74] discuss and outline the specifications of the use of fly ash employed by state departments of transportation in the southern coastal region of the USA. Emphasis is placed on permeability, sulfate resistance, corrosion protection, alkali-silica reaction, and reduction of heat of hydration.

Goni *et al.*[75] investigated the micro-structural changes due to carbonation in fly-ash cement mortars. It was observed that the resistance to carbonation was not affected by the replacement levels of up to 15% but decreased for high fly ash contents.

4. More Ash Utilization

4.1. *Coal Bottom Ash*

Ghafoori and Bucholc[76] studied bottom ash concrete used as partial or total replacement of fine aggregates. They reported that the mixing water requirement increases rapidly when bottom ash is used. Nisnevich[77] used bottom ash and fly ash in lightweight concrete by focusing on the porosity, density, and strength.

4.2. *Rice Husk Ash*

Amer *et al.*[78] reviewed the use of rice husk ash in concrete. To improve the quality of clinker, Singh *et al.*[79] studied the addition of 3% boiler-fired rice husk ash to the black meal of a vertical shaft kiln.

El-Hosiny *et al.*[80] used nitrogen absorption to measure the surface properties Portland cement/rice husk ash pastes. The rice husk ash was obtained at three firing temperatures of 450°C, 700°C, and 1000°C. Higher surface areas were obtained for pastes made from rice husks fired at 450°C and 700°C. The surface area and pore volume results were related to the pore structure of the silica produced in the rice husk ash.

Amer *et al.*[81] studied blended cements made from rice husk ash fired at 450°C and Portland cement. Water demand was increased with the increase of rice husk ash content. Sugita *et al.*[82] designed a semi-industrial prototype furnace to produce a highly reactive and homogeneous rice husk ash. By controlling the burning temperature and grinding, the concrete strength, resistance to acid attack, chloride penetration, and carbonation have been improved.

Chemical analysis of rice husk is shown in Table 1. The composition of the mineral ash is normally within the limits shown in Table 2.

Experiments have shown that alkali-metal halides (such as NaF, NaCl, and KCl) are particularly effective mineralizing agents. It is sufficient, for example, to wet the husks with 0.2 N aqueous NaCl solution. It has also been found that the crystallization of the silicic acid can be delayed if the husks are wetted with HCl before thermal decomposition. Very reactive white rice husk ash can be obtained in this manner at 700°C. It has also been found that the removal of fixed carbon from the ash can be accelerated by adding mineralizing agents such as NaF, NaCl, or KCl. Systematic ashing tests on a semi-industrial scale in the presence of stagnant air have shown that a white ash is obtained in the temperature range between approximately

Table 1. Typical Chemical Composition of Rice Husk.

Properties	Composition in Percent
Water	9
Protein	3.5
Fats	0.5
Cellulose	30–42
Pentosan	14–18
Mineral ash	14–30

Table 2. Typical Chemical Composition of Rice Husk Ash.

Chemical Properties	Composition in Percent
SiO_2	92–97
Al_2O_3	0.75–3
Fe_2O_3	0.17–2
CaO	0.36–3
MgO	0.32–1.5

Table 3. Loss of Weight and Color of Ash During Thermal Decomposition of Rice Husk During Spontaneous Heating to the Temperature Indicated (Duration: 1 h Stagnant Air).

Processing Temperature	Percentage Weight Loss	Color of Ash Residue
300°C	29.8	Black
400°C	66.8	Gray
500°C	77.1	White
600°C	74.7	Gray
700°C	69.4	Black
800°C	73.5	Black
1000°C	75.8	Black

450°C and 550°C, as shown in Table 3. It shows that the organic constituents of the husks are not decomposed to an adequate extent if the temperature is below 450°C to 550°C. Above 550°C, heating-up is spontaneous and the ash is unsuitable (black, carbon-containing). If the temperature is increased far above 1000°C, there is no improvement in the rate of decomposition and the purity of the inorganic end product. The greatest loss of weight occurs during spontaneous heating at approximately 500°C. The end product is white, whereas the quality of the ash progressively decreases above and below 500°C.

4.3. *Palm Oil Fuel Ash*

The pozzolanic properties of palm oil fuel ash, a waste material obtained on burning of palm oil husk and shell, was studied by Hussin *et al.*[83] Compressive strength test with Portland cement substitution levels between 10% and 60% indicate the possibility of replacing 40% ash without affecting concrete strength. A maximum strength gain at the 30% level was achieved.

Awal *et al.*[84] utilized palm oil fuel ash to reduce the expansion of mortar bars containing tuff as a reactive aggregate. According to the results, the palm oil fuel ash has a good potential in suppressing alkali-silica reaction expansions.

4.4. *Other Ashes*

Thomas[85] investigated the possibility of recycling fly ash from the incineration of beef cattle manure. Compressive strength of mixtures decreased with an increase in waste ash content. Ranta[86] discussed the quality of fly ash from pressurized coal gasification with added biofuel. A high level of biofuel may cause the fly ash to exceed its standard values for carbon content. Improvement of the ash quality can be reached through oxidation, classification, agglomeration, and thermal or chemical treatment.

Feng *et al.*[87] studied the use of oil shale ash as cement replacement. At 10% replacement, the compressive strength of the concrete was increased by 5–10% at 28 days. The oil shale ash can also be used as the carrier for a superplasticizer. Kikas[88] reports the intergrinding of a lime-rich oil shale fly ash with Portland clinker to produce a pozzolanic cement. More than five million tons of this cement has already been produced for use in reinforced and prestressed concrete structures.

4.5. *Applications*

Latona *et al.*[89,90] made autoclaved cellular concrete (ACC) from fly ash for environmental purposes. Leachate concentrations were below the regulatory limits. The release of hazardous polyaromatic organic compounds and measured radon emissions from the fly ash were too low to cause potentially hazardous buildups in confined areas. Hu *et al.*[91] studied the strength properties of ACC blocks, made of up to 70% in weight of fly ash. Density influences the compressive, tensile, and flexural strengths. Glasser[92] discussed the mechanisms of immobilization of hazardous elements in cement matrix containing materials such as slag and fly ash. It was argued that since these materials react slowly, their beneficial effects may be underestimated in accelerated short-term leaching tests.

Delsol *et al.*[93] used a sulfatic fly ash produced through a slaked lime desulfurization system for pavement foundations and roller compacted concrete. Papayianni[94] describes the use of a high-calcium fly ash as part of the binder for the roller compacted concrete. The fly ash was treated to reduce its free lime content to below 3%. Advantages of such a material include its low heat generation and slow strength development. Wasserman and Bentur[95] studied the properties of concrete made of sintered fly-ash lightweight aggregate. The strength, absorption, and pozzolanic activity of these aggregates are process dependent.

Cabrera and Al-Hasan[96] used a pozzolanic compound containing 70% Portland cement and 30% fly ash for maintenance, repair, and strengthening of concrete structures. The compressive strength, bond strength, porosity, and permeability were measured. Petzrick[97] used class F fly ash as a substitute for Portland cement in sealing abandoned underground mines. Yu *et al.*[98] reported a reduction in radon emission rates from concrete produced with 25% by weight substitution of fly ash. The reduction occurred for curing periods of 1–7 days, but increased for curing period of 28 days. Man and Yeung[99] studied the compressive strength, radium contents, and radon emission rates of concrete blocks with fly ash substituted for cement. It was found that fly ash increased the radium contents, improved the compressive strength, and decreased the radon emission rates.

Zhang *et al.*[100] used high-volume fly-ash fiber-reinforced shotcrete for an enhanced workability and freeze-thaw resistance. Zhang *et al.*[101] studied glass fiber-reinforced cementitious composites (GFRCC) with a Portland cement-high content fly ash as matrix. The effect of fly ash content, the initial curing time, and accelerated aging on the flexural strength of GFRCC were investigated.

Lee *et al.*[102–112] explored the use of TIA in a variety of blended cement products such as bricks, paver blocks, lightweight palm clinker concrete, foamed concrete engineered shear wall, and foamed grout for soft soil stabilization. Water permeability test system has been developed based on selected European Standards.[113–114] Research products with potential for commercialization are shown in Figs. 1 to 4.

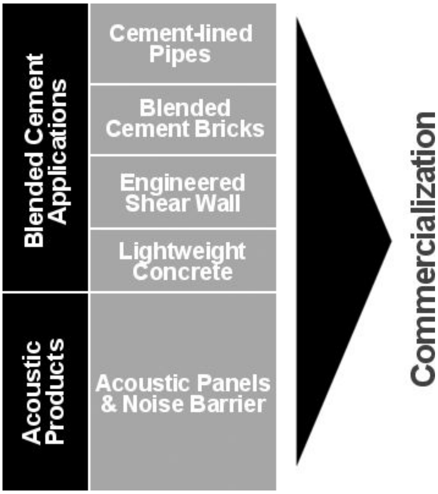

Figure 1. Research Products with Potential for Commercialization.

Figure 2. Blended Cement Bricks.

Figure 3. Research Products with the Use of TIA Blended Cement.

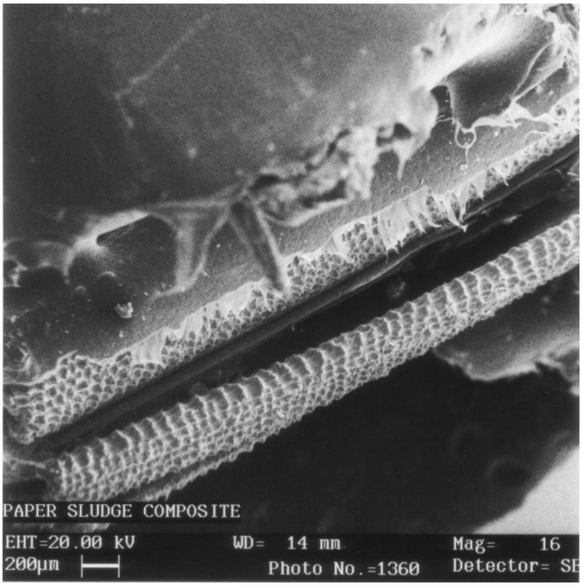

Figure 4. SEM Micrograph of Paper Sludge.

The synthesis and characterization of biomass silica and its effects on construction materials were explored.[115–118] The compressive strength development and water absorption of expanded coal slag (ECS) concrete were presented in the ECS mix design nomograph that can be used to predict the dry density, compressive strength, and water absorption of various percentage of ECS concrete. The structural performance of reinforced concrete beams containing recycled crushed foamed concrete fine aggregate (RFA) was studied. Pure bending cracks were found in RC beams subjected to three points bending test, and combine shear and bending failure cracks for the beams subjected to four points bending test.

The mix design, test method, and performance of controlled density pervious lightweight concrete were studied. The high porosity allows water from precipitation and other sources to pass through it, thereby reducing the surface runoff from going straight into the drain. The recharging of water into designated retention ponds are designed to minimize the fluctuation of groundwater levels. The study is to establish an economical method for construction on problematic soils especially on peat in Sibu, Sarawak. The proprietary lower density foamed aggregate of density between 900 and 1200 kg/m^3 was expected to co-exist with peat. The self-leveling waterproof foamed concrete of density about 1200 kg/m^3 is to be placed on the foamed aggregate as subbase. A special provision for lateral restraint is to be installed to reinforce the subbase and subgrade. Such novel geotechnical structure is to be known as reinforced peat abbreviated as REPEAT. Mechanisms of failure have been identified at the Research Centre for Soft Soil (RECESS, Malaysia), UTHM. The findings are expected to lead to new theory and innovative methods of construction of such problematic soils. The pervious low carbon media with TIA has been studied on its potential for harvested rainwater treatment.

As the cement industry contributes about 5% to global CO_2 emissions from the calcination process of limestone and combustion of fuels in the kiln, a model of CO_2 uptake by biomass silica foamed concrete is proposed as a potential mitigation strategy against CO_2 emission. A comparison between CO_2 emission and CO_2 uptake by carbonation is made. The proposed foamed concrete absorbs CO_2 42.7% faster than the normal concrete, with a regression accuracy of 0.98. Successful deployment could contribute toward sustainable development while benefiting from the carbon credits.

5. Other Solid Wastes

5.1. *Paper Sludge*

Paper sludge is one of the solid wastes produced by paper industry. Paper sludge contains fragments of paper fiber. These fibers are not suitable for use in recycled paper. Disposal options include landfill and energy production. Usually, in the paper production, only 65% of the paper pulp will be turn into paper, the other 35% will be the waste material in the form of paper sludge. Paper sludge contains a high percentage of very fine kaolin clay and fillers, which is used to create a smooth finish on fine paper. Besides kaolin, there are also other contaminants in the paper sludge, such as de-inking compounds, surfactants, residuals from inks and dyes,

Table 4. Chemical Composition of Paper Sludge Obtained from Paper Mill.

Properties	Chemical Composition
pH	7.94
Arsenic (As)	<0.05 ppm
Barium (Ba)	1.2 ppm
Boron (B)	<0.2 ppm
Cadmium (Cd)	0.003 ppm
Chromium (Cr)	<0.001 ppm
Copper (Cu)	<0.01 ppm
Lead (Pb)	<0.05 ppm
Mercury (Hg)	<0.001 ppm
Nickel (Ni)	<0.01 ppm
Selenium (Se)	<0.1 ppm
Silver (Ag)	<0.01 ppm
Tin (Sn)	<0.1 ppm
Zinc (Zn)	0.33 ppm
Chloride (Cl)	0.028%
Oil and grease	2470 ppm
Cyanide	<0.05 ppm

compounds from laser printing, photocopying, and also coagulant that has been used to coagulate the paper sludge during the disperser process. Heavy metals is one of the contaminants exist in paper sludge. Some of these metals are toxic to plants, animals, and also human being. When the paper sludge is dumped in the dumpsite, there are possibilities for leachate from the sludge to get into groundwater, stock ponds, or drinking wells. Besides, when the paper sludge is dumped in the dumpsite, it will undergo anaerobic decomposition and this will cause odor problem. The paper sludge used in this project is obtained from the paper mill located in Kluang, Johor, Malaysia. This paper mill produces 30 metric tonnes of paper sludge per day and the cost for transporting the paper sludge to the dumpsite is approximately RM 10,000 per month. Table 4 summarizes the chemical composition of paper sludge.

5.2. *Rice Husk*

Rice is a basic food in Asian countries such as Malaysia, Thailand, Indonesia, India, and Japan. It is abundantly available in most part of the world. Substantial amounts are also cultivated in various countries in America and Europe. The present world rice production of about 400 million tons per year will probably increase in the future, owing to the great increase in population, particularly in Asian countries. When rice grains are husked, the husks make up about 14–35%, depending on the variety of rice. Since the husks have a low bulk weight of about 100 kg/m^3, they take up 560–1400 million m^3. Although rice husk has its traditional uses, it is mostly under-utilized and causing disposal problem in most countries. It is estimated that 54 million tons of rice husk is produced in China every year. According to a recent study, Malaysia generates about 3.41 million m^3 of rice husk annually with an average density of 100 kg/m^3. Innovative exploitation is crucial to avoid the waste of energy and environmental pollution.

5.3. *Microstructure*

The microstructure of paper sludge was examined using the scanning electron microscope (SEM). Figure 5 shows the SEM micrograph of the paper sludge. It shows the pores and the surface texture of paper sludge. The perforated surface would probably increase the sound absorption coefficient of the paper sludge composite. When the sound wave propagates through the porous absorption material, part of the energy will dissipate by frictional and viscous loss within the pore. The sound wave will also cause vibration to the molecules of the material and the sound energy will be converted to heat energy.

One of the SEM micrographs for rice husk (Fig. 6) clearly shows the shape of the rice husk. The curved shape of rice husk increases the volume of air void in paper sludge composite thus increases its porosity. This will enhance the sound absorption properties of paper sludge composite containing rice husk.

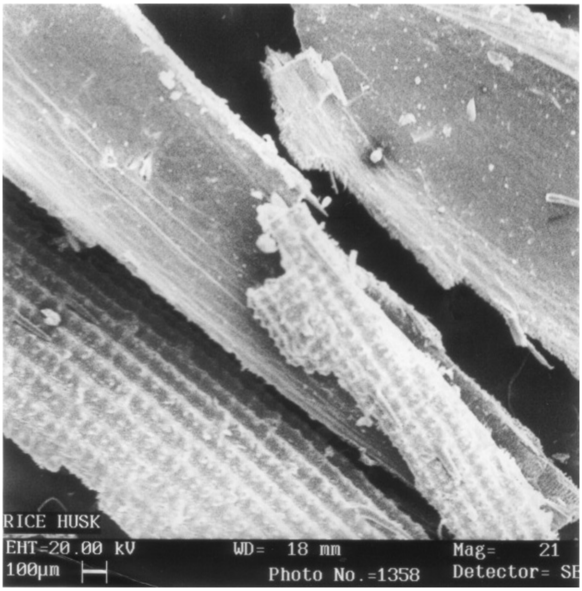

Figure 5. SEM Micrograph of Rice Husk.

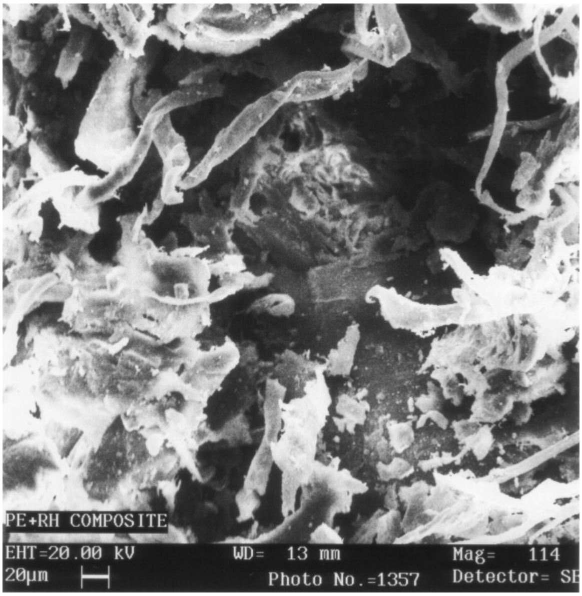

Figure 6. SEM Micrograph I of Paper Sludge Composite.

6. Paper Sludge and Rice Husk Composite

The microstructure of the paper sludge composite (Figs. 7 and 8) shows that there are pores inside the composite. This indicates that the paper sludge composite is a porous material. The porous structure of the composite is expected to enhance the sound absorption properties of the composite. When the sound wave propagates through the porous absorption material, part of the energy will dissipate by frictional and viscous losses within the pore. Therefore, it is believed that the porous structure creates the sound absorption properties of paper sludge composite.

The effect of sample thickness, density, surface texture, and various absorption mechanisms were explored. The microstructure and its physical properties appear to influence the sound absorption coefficient of the paper sludge composite. The properties of the composite in turn rely on the chemical composition of its constituent materials and the processing methods.

The preliminary study indicated that the absorption coefficient of the paper sludge composite varied with different frequencies. Therefore, the absorption coefficient of the paper sludge composite was tested for a wide range of frequencies, ranging from 50 Hz to 6.4 kHz. Peaks were detected at frequencies between 1.0 and 1.5 kHz. The observation of the SEM microstructural images indicated that the porous structure of composite material tends to enhance the absorption properties as generally agreed by acoustics researchers.

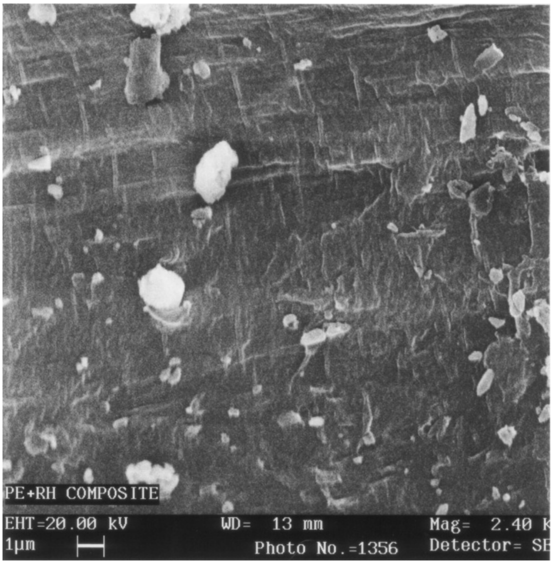

Figure 7. SEM Micrograph II of Paper Sludge Composite.

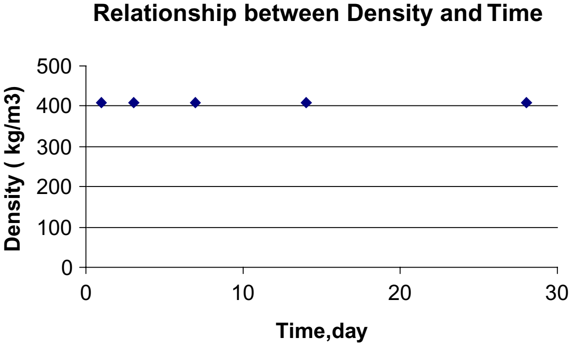

Figure 8. Density Profile.

The discussion is focused on three main areas. First, the sample preparation and the chemical and physical properties of the paper sludge composite are discussed. Second, the chemical and physical properties of constituent materials are analyzed followed by discussions. The final part is the discussion on the effects of sample's density, thickness, surface texture, and surface treatment on its absorption coefficient.

6.1. *Sample Preparation*

As many as 30 large samples and 60 small samples were prepared for the absorption coefficient test. Table 5 shows the material formulation for paper sludge composite.

Table 5. Materials Formulation for Paper Sludge Composite.

Material	Ratio (by weight)
Polyethylene	1
Rice husk	0.3
Paper sludge	0.3

Table 6. Chemical Analysis Data of Paper Sludge Composite Using XRD.

Element	Atomic Percentage, %
Carbon, C	53.24
Nitrogen, N	29.41
Oxygen, O	15.19
Iron, F	2.26
Neon, Ne	−0.10
Total	100

The samples were subjected to a temperature of 150°C. The duration for the heating process differed for the small and large samples. The duration for the heating process is approximately 7 min for the large sample and 5 min for the small sample. The longer heating time is due to the larger mold for the large sample preparation. More heat is needed to be transferred from the hot-press machine to the mold and, therefore, required increased heating duration.

6.2. *Chemical and Physical Properties of Paper Sludge Composite*

The chemical analysis was carried out using the "energy dispersive x-ray florescent spectrometer." The test result for the chemical analysis is given in Table 6. From the table, the test data indicated that the paper sludge composite mostly composed of carbon. A significant amount of nitrogen and oxygen also existed in the composite, at about 29.41% and 15.19%, respectively. The content of iron and neon were not significant and there were no heavy metals in the paper sludge composite.

Physical properties test were carried out to determine the density, compressive strength, and shrinkage of the paper sludge composite. The density analysis is needed to enable the relationship between density and absorption coefficient to be explored. The compressive strength test and shrinkage test were conducted to determine the durability of paper sludge composite.

The average density of the paper sludge composite was 421 kg/m^3. The density profile with respect to time is shown in Fig. 8. The graph shows that there was no change in density with time. The density remained constant for 3, 7, and 28 days. The density of the composite ideally remains unchanged with time because variation in density can affect the absorption coefficient of the composite.

6.3. *Compressive Strength Development*

Compressive strength for the paper sludge composite with an average density of 421 kg/m^3 was determined according to MS 26: Part 2: 1991. The results show that the paper sludge composite has an average compressive strength of 0.67 MPa. Figure 9 shows a plot of compressive strength versus time. The compressive strength remained unchanged with time.

7. Lightweight Concrete

Lightweight concrete-containing biomass aggregate and micronized silica are known as TIA composite. It is a mixture of cement, fine sand, water, and a special foam,

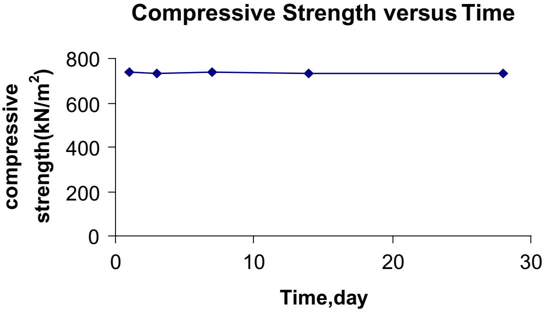

Figure 9. Compressive Strength Profile.

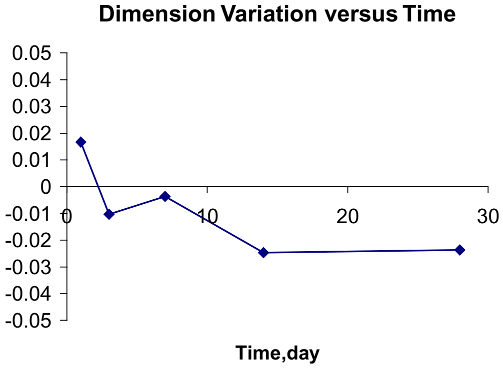

Figure 10. Dimensions Profile.

which once harden results in a strong and lightweight concrete-containing millions of evenly distributed and consistently sized air bubbles. It differs from conventional concrete in that the use of coarse aggregates is eliminated.

The density of TIA composite is determined by the amount of foam added to the cement, sand, and water mixture. TIA composite's density ranges from 250 to 1800 kg/m^3, as compare to 2400 kg/m^3 for conventional concrete. Therefore, the weight of a structure built of TIA composite will be reduced significantly. TIA composite is water-resistant and possesses enhanced sound and thermal insulation properties. Figure 10 shows the dimension profile of TIA composite.

7.1. *Advantages of TIA Composite*

Foam concrete offers several distinct advantages such as the followings:

- Reduces the dead weight of a structure,
- Good workability,
- Can be manufactured to precise specifications of strength and densities,
- Provides enhanced heat and sound insulation,
- Can be nailed, planed, drilled, and sawn,
- Can be applied with all traditional surface finishes — paint, tiles, carpets, etc., and
- Moisture and fire resistant.

7.2. *Savings in Material*

Reduction in dead weights contributes substantially to savings in reinforcing steels in load-bearing structural components such as foundations. The dimensions and, therefore, the overall quantity of steel reinforcement in TIA composite can be reduced by as much as 50%. Savings are also substantial in transportation. No gravel is required to produce TIA composite, only sand, cement, water, and air bubbles are required to produce a foamed mortar. Casting slender walls of 50 mm thickness has been expected to optimize the amount of materials used.

7.3. *Savings in Manpower*

The high flowability of TIA composite makes vibration unnecessary, and thus requires no prescription of vibrating equipment/accessories. Only a few semi-skilled workers are needed to produce TIA composite for casting panels, blocks, and walls for houses. Steelworks, formworks, brick laying, and cement renderings do not constitute major site activities for TIA composite building system. Workers are only needed to set up the reusable formworks, and for the removal of formworks for next erection and casting.

7.4. *Easy Application*

TIA composite is adaptable into existing concrete plants. Foam generator is to produce and discharge foam of a precise dose into the mixer. TIA composite may later be pumped using conventional concrete pumps. Due to the absence of gravel and the ball-bearing effect of the foam, TIA composite possesses a high degree of flowability. No vibration is thus required and TIA composite completely fills all gaps and voids in the concrete or mold, fully embedding any hoses, tubes, and frames for windows and doors.

7.5. *Enhanced Productivity*

The rapid mixing and high fluidity of TIA composite facilitates speedy casting of building elements. With the application of vertical molds to cast complete houses in place, omission of vibrating equipment results in a building being filled in one step including openings for doors and windows. Ducting and conduits for sanitary and electrical services can be cast in place and firmly embedded in the foam concrete.

7.6. *Minimum Cost*

As conventional concrete plants may be used to produce TIA composite, The investment is in the foam generator and the foaming agent — the latter contributes to only about 10% of the overall cost of TIA composite.

7.7. *Acoustic Thermal Insulation*

TIA composite possesses enhanced impact and air-borne sound-absorbing properties that are suitable for partition walls and flooring applications. TIA composite with a density of 1200 kg/m^3, for instance, can produce a thinner monolithic wall that requires lesser raw material. It possesses better insulation properties compared to that of conventional concrete.

7.8. *Foam Generation*

Foam generation requires a foaming agent and a foam generator. Foaming agent is based on a protein hydrolization and is biodegradable. It causes no chemical reaction with the surrounding matrix but serves solely as wrapping material for the air to be encapsulated in the grout. A foam generator is developed from a compressed air

Figure 11. Experimental Foam Generation.

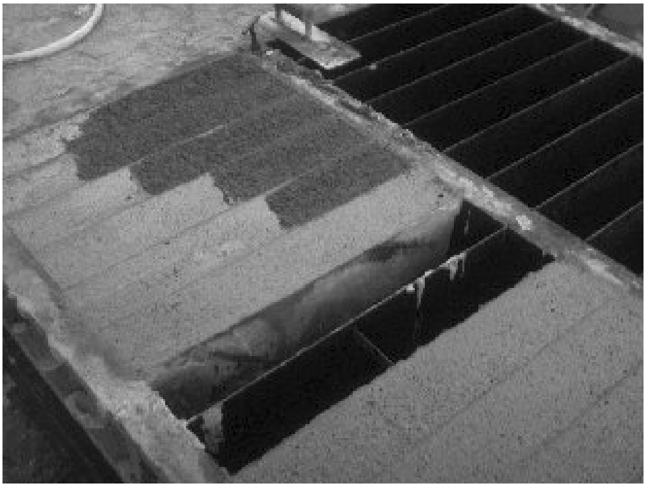

Figure 12. Experimental Production of TIA Composite Blocks.

vessel incorporating a foaming compartment that allows the foaming agent to be agitated by a stream of compressed air. The production of foamed grout is shown in Fig. 11. Figure 12 shows the experimental production of TIA composite blocks.

7.9. *Experimental Foamed TIA Composite*

Fine sand, cement, and water are mixed in accordance with the simplified material formulation for TIA composite of density 1000 kg/m^3 is shown in Table 7.

Table 7. Material Formulation for TIA Composite of Density 1000 kg/m^3.

Material	Quantity
Cement	315 kg
Sand	630 kg
Water	145 liter
Foam	630 liter

Cement and sand are mixed first. Water is then added to the mix to form a slurry. Foam is then added to achieve controlled density. If 1 liter of this mix weighs 1 kg then the density is 1000 kg/m^3 (normal concrete has a density of 2400 kg/m^3). Curing is the same as normal concrete. TIA composite can be demolded after 16 h. TIA composite is recommended to be moist-cured at least for the first two days.

The water/cement ratio must be strictly followed since too little water might cause the cement to draw its requirement from the foam, causing the latter to collapse partly or in total. The foam allows any density of concrete from as low as 250 kg/m^3 to 1800 kg/m^3 to be produced with an optimum ratio of strength-to-density. The possible wide range of densities achievable thus offers multiple and diversified applications, such as site mixing, off-site mixing, prefabrication, pre-cast, or cast in place.

7.10. *Lightweight TIA Composite Panels*

Lightweight TIA composite can be produced in many ways. A modified concrete mixer or a ready mix truck can be used for large capacity mixing job. A small quantity of TIA composite can be mixed with a suitable drum mixer. Production of TIA composite panels to any required size is possible with density ranging from 500 to 1800 kg/m^3. Metal, wood, or plastic mold could be used. Oiling of mold is usually necessary except for PE (Polyethylene) a coated controlled permeability formwork. It is possible to manufacture 300×600 mm TIA composite panels, thickness varies from 75 to 150 mm, with enhanced acoustic thermal insulation properties. Panels with a density of 600–800 kg/m^3 can be produced to a size that can be lifted by a single worker. Curing is essential as cement-based elements need moisture for hydration at an early age. This is particularly true in the presence of direct sunlight which is known to cause rapid dehydration. Curing compound can be applied as an alternative barrier.

7.11. *Effect of Additives*

Mixing randomly distributed short fiber in TIA composite results in a three-dimensionally reinforced matrix. The fiber have the effect of increasing the flexural and impact strength, reducing early-age cracking, and also reducing water absorption. The experimental work on foam concrete containing palm fiber is in progress with members of Malaysia Biomass Industries Confederation (MBIC) *www.biomass.org.my*.

7.12. *Compressive Strength*

An average compressive strength of 2.86 MPa has been achieved for TIA composite cubes with an average density 650 kg/m^3 subjected to 28 days of standard moist-curing. Tests on other densities revealed that 28-day strength exceeding 15 MPa is achievable depending on the density of the mix. Studies also show that compressive strength of above 20 MPa is possible with improved formulation and improvement of admixture.

7.13. *Durability*

TIA composite is expected to have a life span of over 100 years. Previous investigation has shown that the strength of foamed concrete cast 10 years ago continue to increase with age. The TIA composite in many cases provides better alternative to some man-made products such as clay bricks and other construction materials.

7.14. *Quality Control*

TIA composite test specimens are subjected to the dual test system as a quality control measure. Figures 13 and 14 show the testing for water permeability and compressive strength of TIA composite test cubes.

Figure 13. Testing for Water Permeability.

Figure 14. Testing for Compressive Strength.

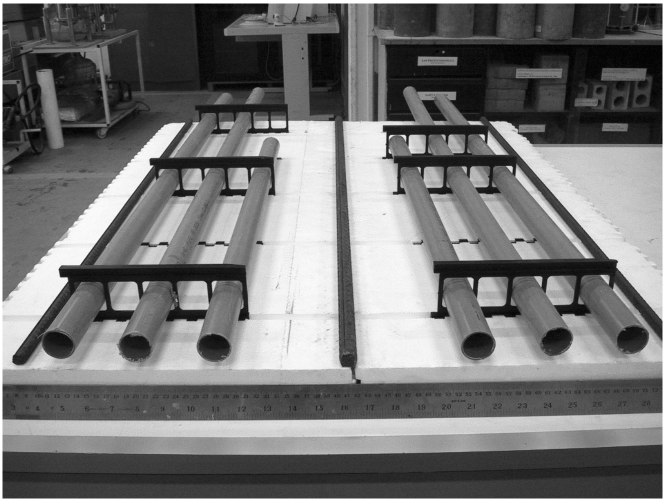

Figure 15. TIA Composite Web-PVC Pipe Reinforcement for Raised Floor System.

Figures 16 shows the typical cross section and the experimental fabrication of the composite panel comprising 75-mm-thick TIA composite cast monolithically to the 50-mm-thick EPS substrate. It is possible to cast panel as thin as 20 mm, short fiber mesh is randomly added to the mix. It can be used as infill for fire-door, acoustic panels, as well as heat insulation-related industries.

Potential applications include insulation work, especially sound proofing applications in roofing and ceiling panels, cavity walls, and as highway noise barriers. Some

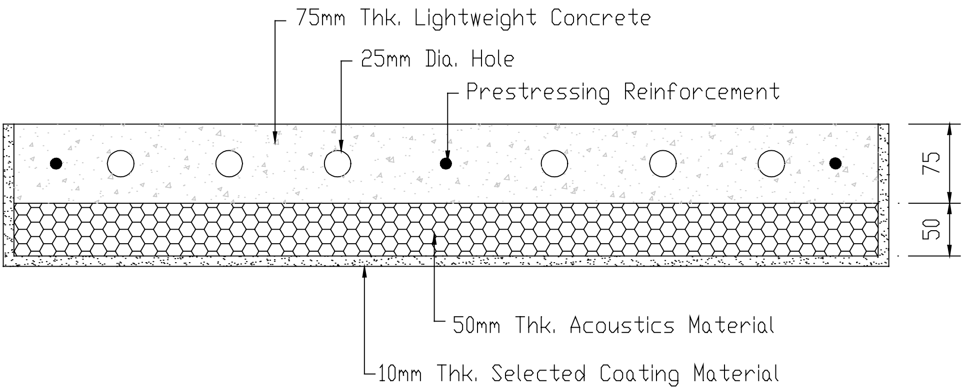

Figure 16. Typical Cross Section of Composite Panel.

experimental acoustic thermal insulation composite panels are shown in Fig. 15 according to the typical cross section of composite panel.

TIA composite is also suitable for special roof deck with enhanced acoustic thermal insulation and water-proofing properties. TIA composite of density between 250 and 550 kg/m^3 is primarily used for thermal insulation or fire protection. It uses only cement (or with a small proportion of sand), water, and foam for insulation board with the same insulating properties as polystyrene, polyurethane, or mineral wool, with no hazardous elements to health and the environment.

8. Effect of Various Properties on Paper Sludge Composite

8.1. *Sound Absorption Coefficient*

A series of experimental studies was carried out in accordance with ASTM E 1050 to examine the effects of density, thickness, surface texture, and various absorption mechanisms on sound absorption coefficient. Figure 17 shows a plot of absorption coefficient versus the frequency for paper sludge composite. The absorption coefficient was found to peak at 1 kHz and decrease gradually after 1 kHz. The absorption coefficient is particularly large at lower frequencies and when the thickness of the samples is equal to $\lambda/4$. Therefore, the optimum frequency may change with different sample thickness.

8.2. *Effect of Density*

The effect on the density of paper sludge composite was determined. The absorption coefficient for the higher density samples, with an average density of 500 kg/m^3, was

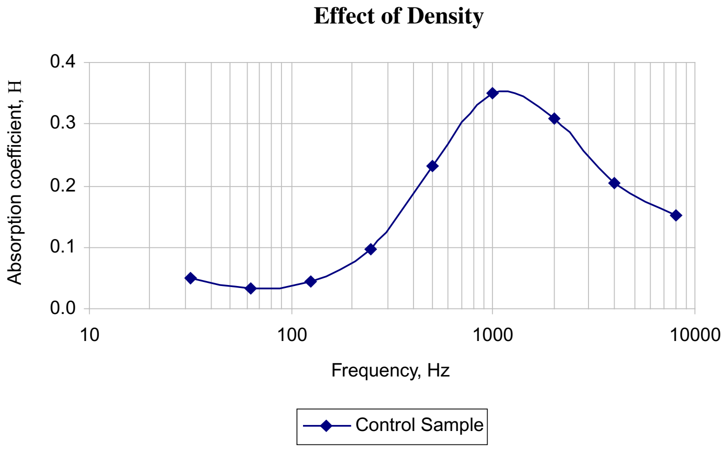

Figure 17. Absorption Coefficient versus Frequency.

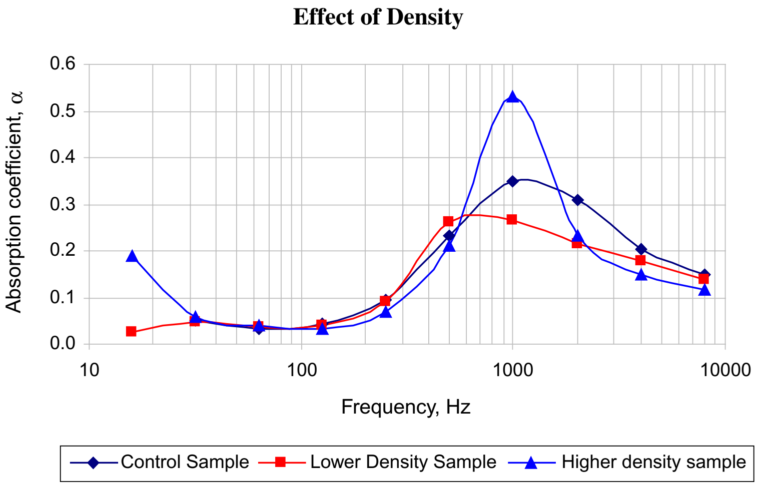

Figure 18. Effect of Density on Sound Absorption Coefficient.

higher than that of the control sample. On the other hand, the absorption coefficient for lower density samples (average density = 370 kg/m^3) appeared to be lower than the control sample. Therefore, the absorption coefficient increased with an increase in the density of composite material.

Variation in density can affect the absorption coefficient as shown in Fig. 18. For the higher density material, the absorption coefficient shows peak at 1 kHz, with a peak α value of 0.53. The graph also shows significant changes in absorption coefficient in frequency range of 300 Hz to 2 kHz. On the other hand, the absorption coefficient of lower-density composite shows peak at a frequency of 500 Hz and

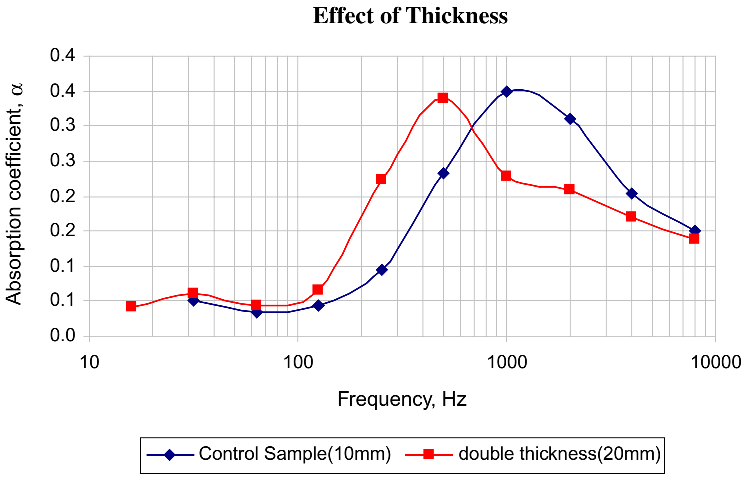

Figure 19. Effect of Thickness on Sound Absorption Coefficient.

with a peak α value of 0.265. From the graph, it can be observed that the optimum frequency shifted from 1 kHz to 500 Hz for lower density material. This indicates that the density tends to affect the absorption coefficient and the optimum frequency of the material.

8.3. *Effect of Thickness*

Figure 19 shows the effect of thickness on the sound absorption coefficient. There is no significant change in sound absorption coefficient between the control sample and the thicker samples.

8.4. *The Effect of Perforation*

The effect of perforated surface was investigated. There are two types of perforated surfaces, they are evenly distributed perforated surface and unevenly distributed perforated surface. Both of the perforated surfaces consisted of 27 holes with a diameter of 3 mm each. Evenly distributed surface means the holes were made on the surface of the sample according to the grid line, where the distance between each holes is the same.

In general, the evenly distributed perforation surface sample demonstrates a higher sound absorption coefficient, with a peak absorption coefficient of 0.98 at 1.5 kHz. Figure 20 shows that there is not much difference between the peak absorption coefficient of the evenly distributed perforated sample and the unevenly distributed sample. The unevenly perforated sample has a peak absorption coefficient of 0.96 at 2 kHz.

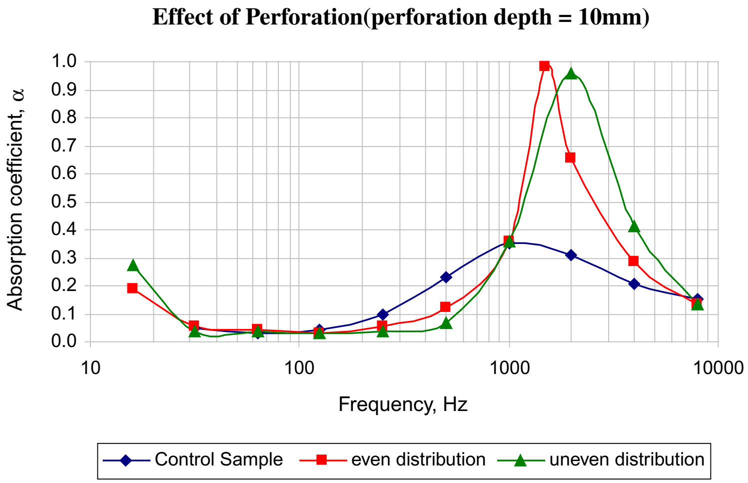

Figure 20. Effect of Perforation on Sound Absorption Coefficient.

In comparison with the control samples, the perforated sample exhibits higher sound absorption coefficient. The underlying reason for this is that the perforated surface creates a resonance effect. When the sound wave arrives perpendicular to the perforated surface, the sound wave will create a large amplitude of air vibration at the opening. Therefore, a part of the energy will dissipate due to the viscous loss. The second reasons for this is that when the sound impinges on the surface of the material, a part of the sound energy will be trapped in holes. This causes frictional and viscous lost in the pores and the energy is converted into heat energy.

8.5. *Effect of Perforation Depth*

Test was carried out to determine the effect of the depth of perforation to the absorption coefficient. The test can be divided into two parts. First, the effect of perforation depth to the evenly perforated surface was determined while for the second part, the effect of perforation depth to the unevenly perforated surface was determined.

It is observed from Fig. 21 that the sound absorption coefficient of the perforated surface is affected by the perforation depth. From Fig. 22, it is observed that the sample with perforation depth of 10 mm exhibited higher absorption coefficient. The absorption coefficient of the 10-mm perforation depth sample was found to be maximum at the frequency of 1.5 kHz, with a peak absorption coefficient value of 0.98. The 5-mm perforation depth sample shows a peak absorption coefficient of 0.93, which is slightly lower than the 10-mm perforation depth sample. From the

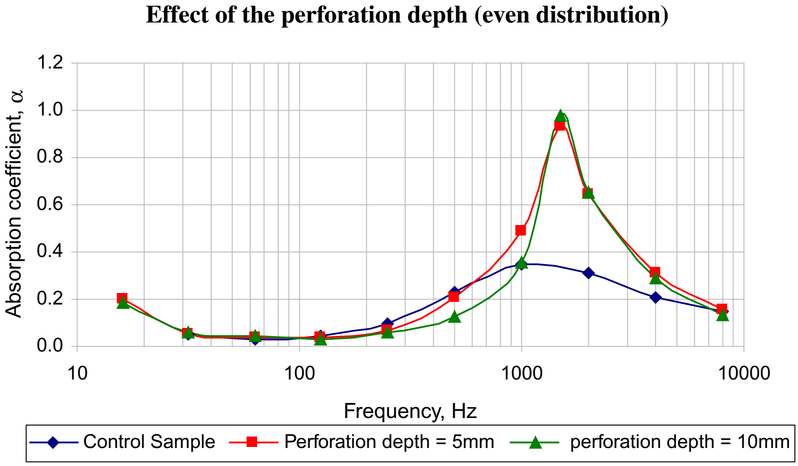

Figure 21. Effect of Perforation Depth on Sound Absorption Coefficient.

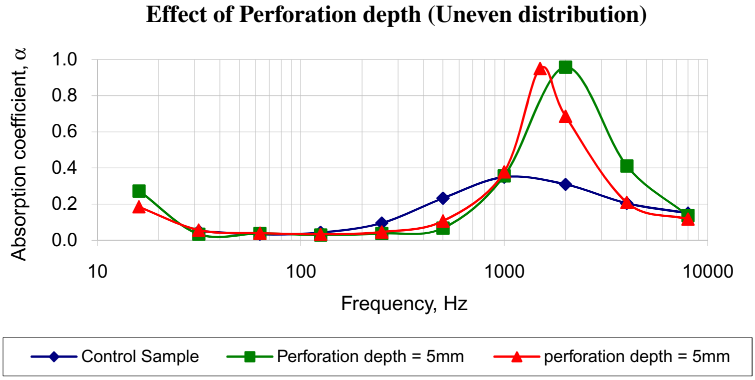

Figure 22. Effect of Perforation Depth on Sound Absorption Coefficient.

graph, it can also be seen clearly that the perforation depth does not affect the peak absorption frequency of the sample. Both of the samples show a peak at 1.5 kHz.

8.6. *Effect of The Depth of Perforations* (*Unevenly Distributed Perforation*)

Figures 21 & 22 shows the effect of perforation depth to the sound absorption coefficient and the peak frequency. The 5-mm perforation depth sample has a peak absorption coefficient of 0.95 at a frequency of 1.5 kHz. From the graph, it is observed that the 10-mm perforation depth sample demonstrates a higher sound absorption

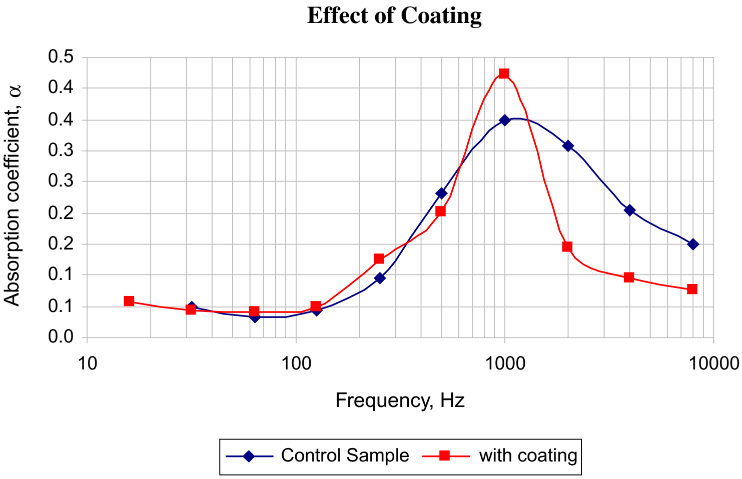

Figure 23. Effect of Coating on Sound Absorption Coefficient.

Figure 24. Fine Sand Mixed with Natural Fibre as Filler.

coefficient. This finding is the same as what has been observed for the evenly perforated sample. However, the 5-mm and 10-mm perforation depth samples have different peak frequencies. This finding is different from the evenly perforated samples where both the evenly perforated samples show peak at a same frequency.

8.7. *Effect of Coating*

Figure 23 shows the effect of coating on the absorption coefficient for control and coated samples. The coated sample exhibits a higher sound absorption coefficient than the control sample, with a peak absorption coefficient of 0.423 at 1 kHz. Therefore, coating does not have adverse effect to the sound absorption coefficient. This result differs from what is claimed by many researchers, where coating will reduce the sound absorption coefficient. As what is claimed by many researchers, the coating will seal the pores on the surface of the absorption material thus reduces the resonance effect.

The findings indicated that the method of application may be one of the factors affecting the test result. The light application acrylic coating may not completely seal the pores on the surface of the composite. Therefore, the effectiveness of the composite to absorb sound will not be very much affected.

9. Acoustic Thermal Insulation Research

This section focuses on the development of acoustic thermal insulation research products and test system with EPS and lightweight cementitious products containing micronized silica. EPS is one of the most versatile and cost-effective cushion packaging materials available. However, most EPS packagings end up as garbage in Malaysia. There are several options for effective recycling of EPS. In UK, EPS accounts for less than 0.1% of municipal waste. Energy recovery is the reclamation of energy, usually in the form of heat from the incineration of waste. When the incinerator conforms to EC regulations for safety and pollution abatement, incinerating EPS is a safe disposal method. EPS has a higher calorific value for heat recovery than coal and could potentially provide a valuable source of energy if incinerated. The study reviews the following to identify areas with good potential for exploitation.

- Energy recovery is one way of generating real value from used packaging materials.
- In a modern incinerator, EPS releases most of its energy as heat, aiding in the burning of other garbage and emitting only carbon dioxide, water vapor, and a trace of nontoxic ash.
- Plastics, such as EPS, in effect “borrow” the energy available from petroleum that is used to produce them and then “return” it when it is used as a fuel in the waste-to-energy conversion.

9.1. *The Recycling Process*

The first step is to separate the waste EPS material into either “contaminated” or clean categories: “Contaminated” EPS could have paper labels, staples, sticky tape, or color dye on it, or been used to carry fish, plants, fruit, or vegetables. Clean EPS has no extra labeling or marks and will have been used to package electrical goods or car parts or similar items. Both clean and contaminated EPS packagings can be recycled, but the process is different. It is vital that the two are separated. Recycling companies will collect EPS packaging for recycling once a sufficient quantity has been stored up. Some will take the packaging exactly as it is, other will want the packaging compacted. Compaction EPS can be compacted to one-fortieth of its original size for easy and cost-effective transportation.

EPS can be extruded and made into a hardwood replacement, which can be used to make products such as garden furniture, window, and picture frames. There is a novel process of converting polystyrene into a replacement wood that is maintenance-free and can be nailed, screwed, or sawn just like wood. Synthetic wood is 100% recyclable. A viable and realistic alternative to hardwood, it makes a contribution to the reduction in deforestation of hardwood forests and it offers a good alternative to landfill as a means of disposal of polystyrene waste. The recycler feeds the compacted blocks of EPS into a granulator that chops the material into smaller pieces. The material is passed into a blender for thorough mixing with similar granules. This material is fed into the extruder, where it is melted.

9.2. *Spray-Applied Coating*

Figures 24 shows the raw materials for spray applied coating, which comprise mainly fine sand mixed with natural fiber and filler, blended cement containing micronized silica as the binder, and the foam to achieve controlled density. Figure 25 shows the blended cement containing micronised silica as binder.

Figure 25. Blended Cement Containing Micronised Silica as Binder.

Figure 26. Foam for Controlled Density Insulation Composite.

Figure 27. Unprocessed Biomass Aggregate (left) and Processed Biomass Aggregate for Lightweight Aggregate (right).

As a continuing effort for sustainable construction, several types of alternative aggregate are experimented such as biomass aggregate. Foam for controlled density insulation composite is shown in Figure 26. Figure 27 shows the processed biomass aggregate as lightweight aggregate.

9.3. *Acoustic Thermal Insulation System*

This project attempts to create value-added acoustic thermal insulation products and test system from recycled EPS. There are several novel features to be incorporated in the composite panel for use in the experimental construction of an acoustic thermal insulation test chamber. The experimental activity is shown in Fig. 29.

The EPS composite panels designed with interlocking joints can be quickly assembled and erected. The web and PVC can be slotted in place with minimum effort as preparation for the construction of the acoustic thermal insulation test chamber. Figure 28 shows the experimental erection of EPS composite panels for floor insulation. The experimental construction of an acoustic thermal insulation test chamber is shown in Figs. 28 and 29 where EPS composite panels are used for wall and floor insulation.

Figure 28. EPS Composite Panels for Floor Insulation.

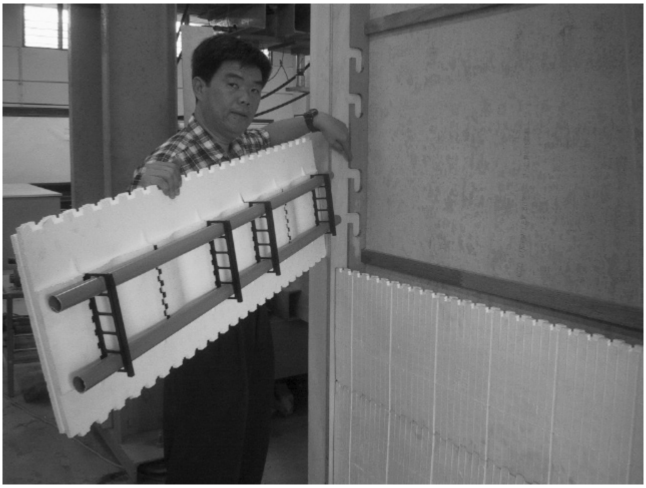

Figure 29. EPS Composite Panels for Wall Insulation.

9.4. *Sound Absorption Coefficient*

The effect of sample thickness, surface texture, perforation distribution, depth of perforation, and surface coating on absorption coefficient were studied. The control sample shows a peak absorption coefficient of 0.35 at 1000 kHz. The increase in thickness does not cause any significant change in absorption coefficient. The effect of perforation was also been observed, where perforation increases the sound absorption coefficient drastically, to a peak value of 0.98 kHz. This is due to the fact that the perforation surface enhances the resonance mechanism of sound absorption. Distribution of perforations on the surface of the paper sludge composite tends to create changes in absorption coefficient and peak frequency. The sound absorption coefficient for evenly distributed perforation sample (distributed according to grid line) is 0.98 at a peak frequency of 1.5 kHz. On the other hand, the unevenly distributed perforation sample exhibits an absorption coefficient of 0.96 at 2 kHz. From the data obtained, it can be observed that even distribution of perforation (holes) on the surface of the samples tends to increase the absorption coefficient and swift the peak frequency to a higher frequency.

The results indicated that the paper sludge composite exhibits peak absorption coefficient at a frequency range of 1–2 kHz, with α ranging from 0.3 to 0.98. With the increase in depth of perforation, the absorption coefficient is observed to increase. The peak frequency remains the same though there is an increase in sound absorption coefficient. The paper sludge composite has an absorption coefficient of 0.35 and 0.98 for perforated sample. The surface coating of the paper sludge composite does not affect the sound absorption coefficient. This is due to the fact that light application of coating does not clog the pores of the paper sludge composite.

10. TIA Composite Building System

A composite building system (shown in Fig. 30) is developed from a combination of lightweight concrete containing micronized silica and recycled EPS insulation panels for ceiling, wall, and floor.

The system offers advantages of economy and speed, comfort, and energy efficiency. The salient features include the use of EPS panel with interlocking profile, webs, and pipes, designed to achieve enhanced acoustic and thermal insulation properties compared to the traditional building system.

11. Engineered Shear Wall System

The engineered shear wall system shown in Figs. 31 & 32 is a novel method of construction. It combines the art of manufacturing a modular system formed from a

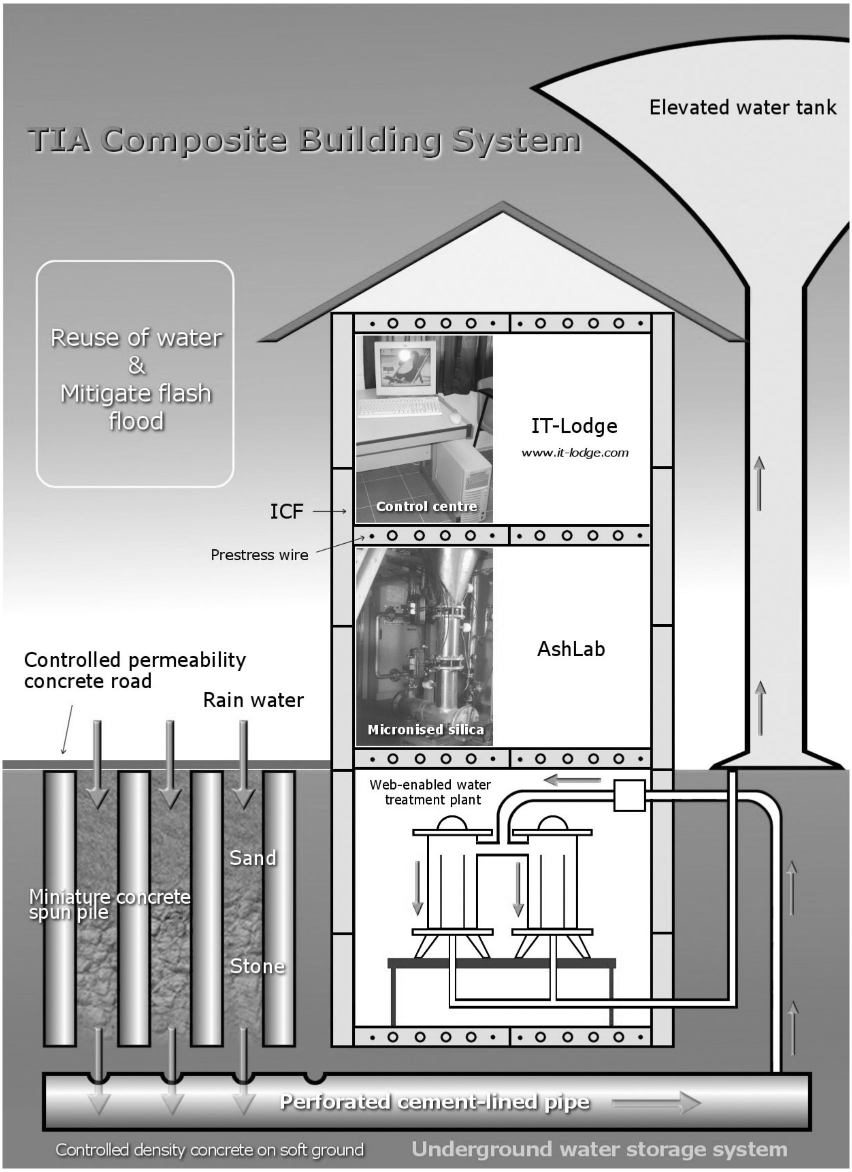

Figure 30. Composite Building System.

self-compacting lightweight reinforced concrete and a method of vertically molding it. The system is aimed at providing a new and improved molded building panel, construction, and connection methods for optimal control of the molding and interconnecting process for such precast building panels, toward enhancing construction safety, quality, and productivity.

Figure 31. Cast *in situ* Engineered Shear Wall.

Figure 32. Expanded Polystyrene (EPS) Panel for Engineered Shear Wall System.

12. Soft Soil Foundation System

A method of improving soft ground containing organic material utilizes a fluid composition for injection into the ground for building foundation. It can also be spray-applied for special rehabilitation works. It may consist of cement, water, fly ash, micronized silica, bentonite, and either sand or a lightweight particles such as volcanic ash. Agricultural and industrial wastes such as wood dust, coal, or incinerator ash, processed waste plastics material, cereal husks, or the like.

The composition is injected at prescribed locations to effect replacement of portions of the soft ground. The composition that solidifies *in situ* can be used in combination with lightweight friction piles to offer the optimum solution for soft soil stabilization.

TIA composite is suitable for foundations on soft soil. Slabs of 600 mm thick can be cast layer by layer to achieve the required level. The web and PVC pipes are provided to accommodate steel bar, wiring conduit, or water pipes. The lightweight concrete composite system is used to form a load-bearing wall. It can be cast to any required size. The load-bearing block with a hollow section infilled with lightweight concrete may be coated with a single a layer of spray-applied coating as finishing.

13. Research Showcase — Micronized Silica

Ten years of intensified research into the synthesis of micronized silica (www.ashmann.com) is beginning to create the impact. It is derived primarily from the innovative exploitation of solid wastes from controlled incineration of — a novel process for the synthesis of micronized silica from controlled incineration of biomass in a rice husk-fired power plant. This process has been initiated in collaboration with a rice mill in Bukit Raya, Kedah, Malaysia. Cementitious composites and concrete products containing micronized silica have attracted much research interest and the award of research grants. A summary of the research showcased is available online via a research achievement portal www.1.net.my. The effective use of information and communication technology enables interested researchers to share useful research knowledge and skill toward exploration and discovery beyond the frontiers of knowledge.

As shown in Figure 33, a rotary furnace is developed to provide controlled incineration of rice husk to study the effect of incineration temperature on the purity of rice husk ash. The system will be integrated with opposed jet mill to produce micronized silica. Figure 34 shows the potential application of micronised silica.

14. Summary — Challenges Ahead

Much attention is focused recently on the commercialization of research products. Despite the fact that researchers are given tax exemption on income such as royalty received from the commercialization of their research products, the success rate of commercialization has been modest in Malaysia. Most researchers need support services from patent agents to draft patent specification and the involvement of successful businessmen in the preparation of winning business plan. There is also a

Figure 33. Rotary Furnace for Controlled Incineration of Rice Husk.

need for researchers to be aware of the mindset of businessmen in order to enhance their negotiation skills.

A novel process for the synthesis of micronized silica has been developed. Exploitation of biomass and industrial wastes for the production of acoustic thermal insulation composite is aimed at achieving sustainable construction. A unified approach by researchers and the practitioners to make good use of the findings is expected to enhance the competitive edge of the relevant industries toward improving the quality of life and conservation of the environment.

A method of soft soil stabilization with foamed concrete containing micronized silica and a lightweight composition with solid waste material such as TIA, powdered waste plastics material, rice husk, and other locally available biomass are being experimented. The composition is injected at prescribed locations with innovative precast lightweight hollow-core concrete friction pile to effect replacement of portions of the soft soil. The composition that solidifies *in situ* is expected to provide cost-effective geotechnical solution, as the surrounding soft ground is stabilized. It is envisaged that such blended cement products are competitive and have potential to expedite the physical development of the university. However, the development of blended cement for waterproofing applications has encountered obstacle due the high cost of the opposed jet mill.

Innovative foundation system with the utilization of used tires is being experimented in some housing projects. The engineered shear wall system that is expected to revolutionize the construction process is in the final stage of intellectual property protection and negotiation for commercialization.

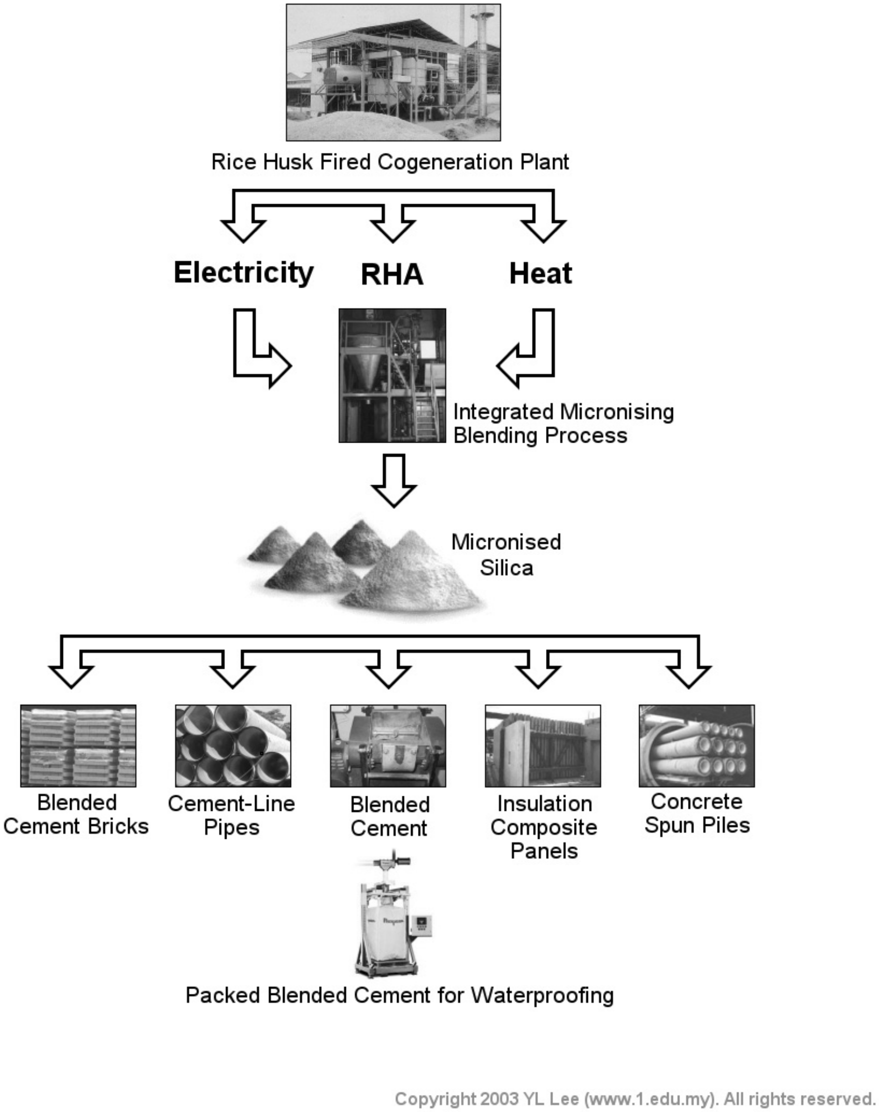

Figure 34. Potential Applications of Micronised Silica.

Blended cement for waterproofing applications and controlled permeability concrete with permeable lightweight concrete formwork has been identified to be the research focus for advanced materials. The proposed project involves materials and test specimens preparation and testing according to ISO/DIS 7031. The effect of miconized silica, paper sludge, and natural fiber on water absorption of lightweight concrete formwork will be the focus of the study.

Acknowledgment

The authors wish to acknowledge the fundamental research grant (Vot No. 84211) awarded by the Ministry of Education Malaysia. Special appreciation to Ban Heng Bee Rice Mill (1952) Sdn. Bhd. (www.malaysiarice.com) and Evergreen Fibreboard Berhad for the contribution of raw materials and Your Website Solution Sdn. Bhd. (www.1.com.my) for the maintenance of the concrete innovation blog www.1.net.my and ash utilization technology portal (www.ashmann.com).

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Chapter 12

THE USE OF BIOLOGICAL METHODS FOR TOXICITY EVALUATION OF WASTES AND WASTE-AMENDED SOILS

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Abstract

Wastes and waste-amended soils can content different contaminants in them. Total concentrations of these contaminants can not be used for ecological risk assessment. Biological methods appeared to be necessary to evaluate the potential ecological impact. Estimation of toxicity of wasts and waste-amended soils plays an important role for decision making concerning further management of them. Nowadays, vary methods of bioindication and bio-testing based on bacteria, crustacean, protozoan, rotifer, alga, higher plant and other organisms are used. In this chapter, the most common used methods are presented.

Keywords: Bioassay, bio-testing, bioindication, elutriate, bioassay, contact bioassay, eco-toxicity.

1. Introduction

Growing anthropogenic load on ecosystems demands the development of measures for minimizing of influences and environment restoration. For effective environmental protection, the exact information about environmental compartments — aquatic, air, soil ecosystems, etc. — is needed. In recent years, the assessment of natural and anthropogenic objects by means of biological methods has received considerable attention.[1] Even if chemical measurements are more and more versatile, accurate and sensitive, they do not provide direct information regarding the biological effect of toxic compounds, nor about the available concentration for biodegradation.[2] Using biological methods, the presence of toxic compounds and their ecological risks can be determined.[3]

The main problem of for biological analysis is the presence of solid objects such as soils and wastes. Due to rapid increases in urban population, municipal solid waste (MSW) and sewage sludge (SS) have increased dramatically in the past 20 years. Environmental pollution caused by MSW and SS has become a serious social problem that hinders urban development, especially for large cities. Increasing concern over problems of environmental pollution and disturbance has stimulated research to evaluate the possible impact of these stresses on ecosystem biota.

Soil is a complex microhabitat regulating plant productivity and the maintenance of biogeochemical cycles by the activity of microorganisms able to degrade organic compounds including xenobiotics. Several bioindicators of soil health and quality have been developed and reviewed. Among them, microorganisms, due to their capability to respond quickly to environmental changes, are expected to be efficient bioindicators.

For their toxicity estimation, different methods are used, e.g. methods based on microorganisms.[4,5] The importance of microbial activity in cycling organic matter and regulating active nutrient pools suggests that the effects of pollution on microorganisms are fundamentally related to the effects on crops, natural vegetation, and ecosystem productivity. Microbial activity, therefore, appears as good indicators of the degree of pollution of contaminated ecosystems.[3]

1.1. *Special Terms That are Used in Biological Method*

Test — The procedure of toxicity estimation of the sample by means of bioindication method or bioassay.

Bioindication method — Biological method that is based on the estimation of changes of parameters (growth rate, biomass, microbial enzyme activity, etc.) of sensitive organisms of native flora and fauna.

Bioindicator — Sensitive species of native flora and fauna that are used to indicate pollution or other impact on ecosystem.

Bioassay — Biological method that is based on the estimation of changes of parameters (growth rate, biomass, microbial enzyme activity, etc.) of organisms that are introduced into the tested sample.

Test object — Organism that is introduces into the testing sample in bioassay, whose parameters (growth rate, biomass, microbial enzyme activity, etc.) are measured to estimate the presence of pollutants or other impact of the tested sample.

Test function — Parameters (growth rate, biomass, microbial enzyme activity, etc.) of test object or bioindicator that are measured in bioassay or bioindication method.

Toxicity index — Value that expresses toxicity of the tested sample, e.g. EC_{50} — the concentration of the sample that cases 50% inhibition of the test function.

2. Bioindication and Biotesting

All the methods of biological analysis can be classified in two main groups — bioindication and biotesting. Differences between the two groups are demonstrated in Table 1. In bioindication, the characteristics of activity, number, or biomass of indigenous organisms by the influence of toxicants are measured under native or laboratory conditions. During the biotesting, the laboratory-cultivated organisms are brought into the contact with the sample and their response is registered.

The most developed methods of waste-amended soils are methods of bioindication (Fig. 1). These methods are hardly used for pure wastes. In these methods, the characteristics of native soil organisms — plants, arthropods, etc. — are measured.

Many of soil bioindication methods, such as measurements of total soil biomass, microbial diversity, enzymes activities, and oxygen uptake rates, have been developed for microorganisms. The main bioindication tests based on microorganisms are described below. Bioindication methods are widespread because of their

Table 1. The Two Main Groups of Methods of Biological Analysis.[8]

Category of Methods	Category Description	Toxicants	Organisms	Conditions
Bioindication	Assessment of polluted sample using sample organisms	+	+	+/−
Biotesting	Assessment of polluted sample using introduced pure organism culture	+	−	−

(+) Means taken from tested sample and (−) means not taken from tested sample.

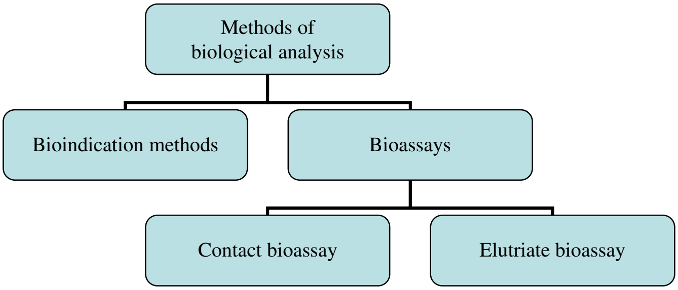

Figure 1. Classification of Methods of Biological Analysis.

sensitivity and rapid answer to pollution or other change in the environment. However, by using of bioindication methods, the problems of the interpretation of results occur. To estimate the intensity of anthropogenic impact, the information about organism — bioindicator — without the impact (control sample) is needed. The control sample must have the same characteristics as testing sample exclusive pollution or other impact. The absence of control sample is the main disadvantage of bioindication methods. Besides, bioindication has the following disadvantage. Most of the soil microbial parameters are integrals, which means that one parameter reflects metabolism processes of two or more species. Therefore, even if one of the species, which is the most sensitive one, is inhibited, other species will retain the value of the integral the parameter stable. For instance, the parameter "total microbial biomass" is a sum of biomasses of many microbial species presented in the sample, such as bacteria, archea, fungi, and actinomycetes. The presence of toxicants influences different microbial groups differently. Biomass of first group can increase, of the second group can remain the same, of the third group — decrease. Therefore, the total microbial biomass being an important environmental relevant index, can be insensitive to pollutants because of the combination of stimulating and inhibiting effects on different members of microbial community.

Besides, the results obtained using bioindication methods can depend on several random factors, such as weather conditions (temperature and humidity) at the moment of sampling. Possible adaptation of the native communities in contaminated soils may induce a significant bias in this evaluation,[6] which is another disadvantage of bioindication methods.

Simultaneous measurement of many soil parameters (e.g. activities of many soil enzymes) might be more valid for estimating the overall response to diffuse pollution and environmental stress than the determination of the single parameter.[3]

In contrast to bioindication methods, in biotesting, the procedure of testing of the natural control samples is not needed. Extracting water or culture broth can be used as a control. In bioassays, genetic homogeneous laboratory population and stable

Table 2. Commercial Tests for Toxicity Estimation.[12]

Test Name	Test Object and Test Function
BioTox Kit	*Vibrio fischeri*, luminescence inhibition
Microtox Solid-Phase Test	*V. fischeri*, luminescence inhibition
ECHA Biocide Monitor	*Bacillus* sp., dehydrogenase activity inhibition
MetPAD	*E. coli* mutant culture, β-galactosidase activity inhibition
MetPLATE Kit	*E. coli*, β-galactosidase activity inhibition
Toxi-Chromotest Kit	*E. coli* mutant culture, β-galactosidase activity inhibition
MetSoil	*E. coli* mutant culture, β-galactosidase activity inhibition
Toxi-ChromoPad Kit	*E. coli*, inhibition of *de-novo* β-galactosidase synthesis
Polytox	Mixed bacterial culture, extracted from wastewater, respiration activity inhibition
Muta-Chromoplate Kit	Modified Aims test
Mutatox	Dark mutant culture of *Photobacterium phosphoreum* (*V. fischeri*), genotoxicity
SOS–Chromotest KitE	Mutant culture of *E. coli*, genotoxicity
Daphnotoxlit F magna	*D. magna*
Daphnotoxkit F pulex	*D. pulex*
IQ Toxicity Test Kit	*D. magna*
Artoxkit F	*Artemia franciscana*
Thamnotoxkit F	*Thamnocephalus platyurus*
Rotoxkit F	*Branchionus calyciflorus*
Protoxkit F	*Tetrachymena thermophila*
Algaltoxkit F	*Selenastrum capricornutum*

experiment conditions are provided. The influence is estimated as an effect of the sample on the laboratory test object. That is why results obtained in different times and places are comparable. Bioassays can be used both for waste and waste-amended soil toxicity analysis. Moreover, by means of bioassays, the hazardous classes for environment of wastes can be determined. The main disadvantage of bioassays is their possible irrelevance to the real toxicity of soils and wastes because of many experimental steps and problem of the extrapolation.

Nowadays, a lot of bioassays inclusive commercial tests are developed (Table 2).

The majority of bioassays for testing of solid samples have been performed on water or solvent extract (elutriate). Such tests are, therefore, called elutriate ones. Elutriation makes investigation simpler.

The main test functions such as growth rate, cell viability, or inhibition if various biochemical properties of cells are easily measured. The authors of elutriate methods assume that only water dissolved compounds affect the organisms and that results obtained on elutriates reflect the real toxicity of the solid sample. However, it is difficult to provide an interacting system of contaminated solids and bacteria, while retaining the ability to recover them for toxicity measurements. Therefore, toxicants are almost always extracted from the solid samples and presented to test object in liquid form. This leads to a fractionation of the toxicity, which may underestimate or overestimate exposure routes and consequently potential adverse environmental effects.[7] Elutriate bioassays have several disadvantages:

- They can underestimate the type, bioavailability, and quantity of pollutants.
- Elutriate toxicity indexes cannot be directly extrapolated on solid samples, even by using experimental data of ration between soil and soil solution of different elements.
- The most important role for bioavailability of some chemical elements for some test objects has the way of exposition, because even the combined toxicant can influence the test objects.[6]

A bioassay with direct contact to solids can integrate the real situation in a more complex system much better that extracts.[7] Many methods of contact bioassay based on plans and soil invertebrates are developed. Less contact bioassays based on microorganisms are described.

3. Choice of the Appropriate Method of Biological Analysis

Both bioindication and biotesting methods have their advantages and disadvantages. The two groups of methods should, therefore, be used in parallel to provide complementary information concerning the level of perturbation in ecosystems.[3]

A selection of the appropriate biological method for measuring soil toxicity is complicated. When selecting test organisms or bioindicators, we should choose species that are representative of functional roles played by resident organisms, and sensitive to the contaminants present. In addition, test species of greatest utility will have the following characteristics: rapid life cycles, uniform reproduction and growth, ease of culturing and maintenance in the laboratory, uniformity of population-wide phenotypic characteristics, and similar routes of exposure to that which is encountered in the field. However, increasing evidence suggests that the use of a single bioassay will never provide a full picture of the quality of the environment. Therefore, a set of tests composed of bioassays on different trophic levels is necessary to provide an accurate assessment of environmental quality.[8]

In many studies on solid wastes in which ecotoxicological tests have been used, little attention has been paid to such aspects as the selection of test species, the sensitivity of tests, and the simplicity and cost of the tests. Very few serious endeavors have been made to determine which is the minimum battery of tests needed. A battery of toxicity tests should be used to evaluate the effects on different trophic levels, as well as acute, chronic, and genotoxic effects. Various single-species screening tests are often employed to detect possible harmful effects of chemicals on biotic systems. In the environment, however, different species and a complex food web are exposed. Various bacterial bioassays have been developed for the screening of chemicals, effluents, sediments, and soil toxicity. Microbial tests (e.g., MetPLATE, ToxiChromotest, Microtox, Mutatox, BioTox) are simple, rapid, sensitive, and inexpensive toxicity tests. These tests are based on the measurement of enzyme activity or metabolism of bacterial cells. Mitochondrial enzymes and enzyme reactions, including mitochondrial electron transport (reverse electron transport, RET), are well-understood processes in living organisms. Being fast and inexpensive, mitochondrial enzyme reactions have the potential to be sensitive and efficient screening tests for general toxicity. The RET assay has been suggested as an alternative to Microtox for screening water-soluble chemicals. Inhibition of RET has been found to correlate well with whole-animal toxicity in aquatic organisms and man. Soil animals are relatively vulnerable to the adverse effects of oil contamination. Soil animal community responses to presence of light fuel oil and hydraulic oil. Soil arthropods and enchytraeid worms were especially sensitive to oil spillage. Soil animals have been used as bioassays in several studies. Earthworms are the most common, but collembolans have also been used. Several species of higher plants have been used to test phytotoxicity. Terrestrial plant tests, seed germination and root elongation tests, may reflect both direct and indirect (water-mediated) effects of the soil. In general, plant tests are cost-effective, relatively easy to perform, and are suitable for turbid sample testing.[9]

Genotoxicity tests are another important group of bioassays. Many assays to assess DNA damaging ability, mutagenicity, or clastogenicity are developed and published. These include popular bacterial and plant assays such as the *Salmonella* mutagenicity test, the anaphase aberration test in *Allium cepa*, and the *Tradescantia micronucleus* test, as well less common tests such as micronucleus induction test in *Xenopus laevis* and rifampicin resistance forward mutation test in *Pseudomonas putida*.[10]

Both contact and elutriate bioassays can be classified according to the test function and test object that has been chosen. According to the test object, biotests are divided into seven main groups. Each group differs by test functions and duration of the experiment (Table 3).

4. Expression of Toxicity

Toxicity of wastes and waste-amended soils can be expressed by different indexes. In most cases, these indexes are calculated on the basis of dose-response curves. Dose-axis (abscissa), which evaluates the influence (e.g. concentration, time of influence, etc.) can be plotted in linear or logarithmic scale. Response-axis (ordinate) evaluates

Table 3. Biotests Test-object Classification.[7]

Bioassay Group and Example of the Test Object	Contact or Elutriate Test	Type of Test	Test Function	Duration
Cell organelles				
RET assay, beef heart submitochondrial particles	Elutriate	*In vitro* test, enzyme complex	Enzyme inhibition	20 min
Microorganisms				
BioTox, *V. fischeri*	Elutriate	Microbiotest	Luminescence	5–30 min
Bioluminescent direct contact test, *V. fischeri*	Contact	Microbiotest	Luminescence	5–30 min
Toxichromotest, *E. coli*	Elutriate	Microbiotest	Enzyme inhibition	6 h
MetPLATE, *E. coli*	Elutriate	Microbiotest	Enzyme inhibition	4–5 h
Bacterial growth inhibition test, *Pseudomonas putida*	Elutriate	Microbiotest	Growth inhibition	16 h
Terrestrial plants				
Seed germination test, red clover Duckweed, *Lemna minor*	Elutriate	Plant test	Growth inhibition	4 or 7 days
Root elongation contact test, rape (*Brassica napus*)	Contact	Plant test	Growth inhibition	3 weeks
Algae				
Algatoxkit, *Selenastrum capricornutum*	Elutriate	Algal test		
Soil animals				
Soil arthropod, *Folsomia candida*	Contact	Soil animal test	Survival and reproduction	14 and 32 days
Enchytraeid worm, *Enchytraeus albidus*	Contact	Soil animal test	Survival and reproduction	14 and 32 days
Genotoxicity Mutatox, genotoxicity assay, *V. fischeri*	Elutriate	Microbiotest	Luminescence	24 h
Aquatic animals				
D. magna immobilization test	Elutriate	Aquatic animal test	Mortality (immobilization)	48 h

alteration of the test function by growing influence. Often, inhibition or relevant activity of the test function is plotted on the y-axis. In Fig. 2, the typical dose-response curve and the method of calculation of one of the toxicity indexes are demonstrated. Several toxicity indexes can be calculated from the curve. The most

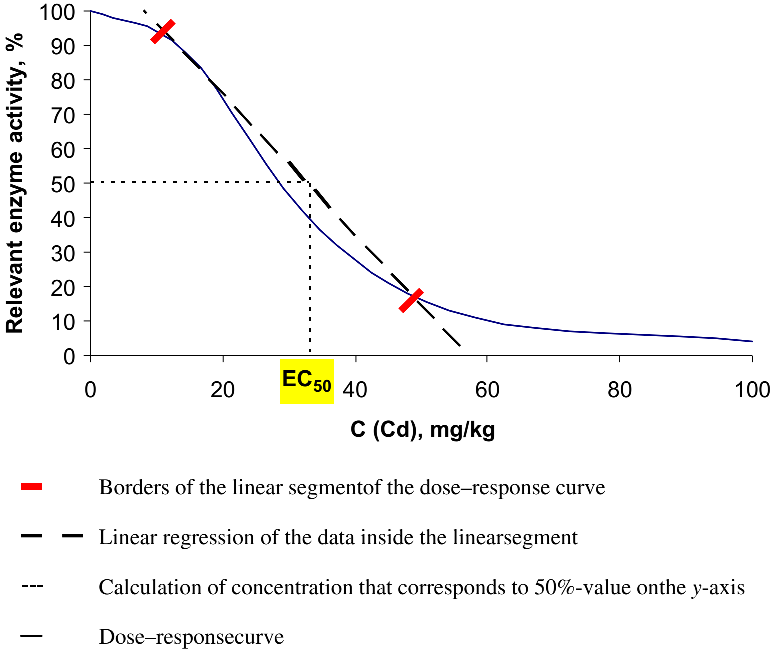

Figure 2. Typical Dose-response Curve and Method of Calculation of EC_{50}.

common index is EC_{50} — effective concentration 50 — concentration that causes 50% inhibition of the test function. For calculation of EC_{50}, the linear segment of dose-response curve, which includes 50% of *y*-value, is chosen. Using data of dose and response inside the chosen segment, the linear regression is modeled. From the model, value on *x*-axis that corresponds to 50%-value on *y*-axis is calculated.

In similar way, EC_{10}, EC_{20}, EC_{80}, etc. and LC_{50} (lethal concentration), LD_{50} (lethal dose), or ED_{50} (effective dose) are calculated. Toxicity of soils can be expressed as soil toxicity units ($STU = 1/EC^{*}_{50}1{,}000$). Besides, toxicity can be expressed as LID_{10}. Waste elutriate is diluted with different factors, these factors are plotted on the *x*-axis. LID_{10} expresses dilution factor of waste elutriate that causes 10% inhibition of a selected biological response. There are many other toxicity indexes that are used in the special literature:

NOEC — no observed effect concentration,
NOAEC — no-observed adverse-effect concentration,[11]
PNEC — predicted no effect concentration,
HQ — hazard quotient, and
PEC — predicted environmental concentration.

5. Acute and Chronic Toxicity

A toxicity test may measure either acute or chronic toxicity. Acute responses are expressed as LC_{50} or EC_{50} values, which means that half of the organisms die or a specific change occurs in their normal behavior. Acute toxicity covers only a relatively short period of the life cycle of the organism. For example, tests with daphnia, rats, or birds lasting 24–28 h are considered to express acute toxicity. However, unicellular organisms with a life cycle shorter than 24 h can be used to study chronic toxicity if exposure to the chemical causes the prevention of growth and multiplication. Chronic toxicity tests are used to assess long-lasting effects that do not result in death. Studies encompassing the whole life cycle of the organism or the effects on genotype are examples of chronic test.[12]

6. Bioindication Methods

In this section, microbial methods of bioindication are described. Microorganisms are used because of the reasons described below.

Microbial testing of soil is used when the wastes in particular MSWs and SS are disposed on it. The agricultural use of organic wastes is permitted, since it contains organic matter and is rich in macro- and micronutrients. However, these wastes especially SS and MSWs also contain contaminants, such as heavy metals, organic compounds, and human pathogens. Therefore, the disposal of these wastes on land can either stimulate soil microbial activity, due to an increase in available carbon and nutrients, or inhibit activity, due to the presence of heavy metals and other pollutants. In the case of high content of toxicants in wastes, its application can lead to a possible loss of habitat function of the soil.[6,13,14]

The importance of microbial activity in cycling organic matter and regulating active nutrient pools in soils suggests that the effects of pollution on soil microorganisms are fundamentally related to the effects on crops, natural vegetation, and ecosystem productivity.[15–17] Microbial activity measurements, therefore, appear as good indicators of the degree of pollution of contaminated soils.[18–21]

Microbiological and biochemical properties are very responsive and provide immediate and precise information on small changes occurring in soil.[21–23] There is increasing evidence that such parameters are also sensitive indicators of ecology stress suffered by a soil and its recovery, since microbial activity has a direct influence on the stability and fertility of ecosystems.[24]

Microbial indicator has been defined as a microbial parameter that represents properties of the environment or impacts to the environment, which can be interpreted beyond the information that the measured or observed parameter represents itself. The ideal soil microbiological and biochemical indicator to determine the effects of the wastes would be simple to measure, should work equally well in all environments, and reliably reveal which problems existed where. It is unlikely that a sole ideal indicator can be defined with a single measure because of the multitude of microbiological components and biochemical pathways. Therefore, a minimum data set is frequently applied.[25]

As the soil microbial community plays a major role in soil functioning, in particular C turnover and other nutrient transformations, it is, therefore, important to understand any potential negative effects of the wastes. Microbial biomass, respiration, the metabolic quotient (qCO_2), and soil enzymatic activity can be utilized as indicator for changes in soil quality.[6,26–31] Currently, there is a range of methods available to measure microbial biomass and microbial activity in soils.

6.1. *Soil Microbial Biomass*

Microbial biomass content is an integrative signal of the microbial significance in soils because it is one of the few fractions of soil organic matter that is biologically meaningful, sensitive to management or pollution and finally measurable.[32] As a susceptible soil component, the biomass may be, therefore, a useful indicator since pollution may reduce this pool.

The soil microbial biomass can be defined as organisms living in soil that are generally smaller than approximately 10 μm.[25] The proportion of carbon biomass in the soil has been found to be 1–3% of organic carbon.[33,34] Most attention is given to fungi and bacteria because (a) these two groups of microbes being the most important with reference to energy flow and nutrient transfer in terrestrial ecosystems and (b) fungi and bacteria are generally dominating within the biomass. The microbial biomass consists of dormant and metabolically active organisms. However, the presently widespread biomass estimates, either direct or indirect (biochemical) techniques, were not properly valid and checked for separating these fractions. The proportion of the metabolically active biomass is about 1–5 and 2–8% of the organic matter in arable and grassland soils.

Four indirect methods are most often used now: fumigation–extraction (FE), substrate–induced respiration (SIR), fumigation–incubation (FI), and ATP content. All these methods are designed to quantify the microbial biomass carbon in different soil samples, soil horizons, soil profiles, and sites.[30,35–45]

6.1.1. *Biomass-C by FE Technique*[46]

Soil samples are fumigated with chloroform and extracted with potassium sulfate solution. The organic carbon in the filtrate is determined and converted to biomass C.

6.1.1.1. Procedure

Weigh 150–200 g of field-moist soil into each of four glass vials or wide beakers. Place two of the glass vials and glass beaker, fill with 50 ml of ethanol-free chloroform and anti-bumping granules, into a desiccator lined with moist filter paper (samples). Evacuate the desiccator with water jet pump until the chloroform begins to boil, then close the tap of desiccator.

After 24 h of fumigation at room temperature in the dark, remove the fumigants by repeated evacuation. Ten evacuations are usually sufficient to remove the chloroform. Mix the fumigated soil (one part) with 0.5 M potassium sulfate solution (five parts), shake for 30 min, and filter (sample, make three replicates). Repeat the same procedure but without fumigation for the controls (three replicates). Analyze the filtrates for organic C.

In a round-bottom flask, mix 5–20 ml of filtrate with 10 ml of 0.05 M potassium dichromate solution and 10 ml of sulfuric acid with silver sulfate (1%). Connect the flask to a condenser and boil for 30 min at 150°C under reflux, then allow to cool to room temperature. Remove the condenser, add 100 ml of distilled water and 0.3 ml of indicator solution, and titrate with ferrous ammonium sulfate solution until the color changes from yellow-green over blue-green to reddish brown. Treat controls and reagent blank in the same way.

6.1.1.2. Calculation of results

$$\text{mg organic C g}^{-1}\text{dm} = \frac{(\text{B} - \text{SorC}) \cdot 0.3 \cdot \text{V} \cdot 1000 \cdot 100}{\text{ml} \cdot \text{SW} \cdot \%\text{dm}},$$

where B — mean volume of $(NH_4)_2FeSO_4 \bullet 6H_2O$ consumed by reagent blanks (ml); S — mean volume of $(NH_4)_2FeSO_4 \bullet 6H_2O$ consumed; by samples (ml); C — mean volume of $(NH_4)_2FeSO_4 \bullet 6H_2O$ consumed by controls (ml); 0.3 — conversion factor (1 ml of 0.1 M $(NH_4)_2FeSO_4 \bullet 6\,H_2O$ corresponds to 0.3 mg C); V — volume of extract (ml); 1,000 — conversion factor; ml — aliquot of filtrate; SW initial soil weight (g); and $100 \bullet \%\ \text{dm}^{-1}$ — factor for soil dry matter.

$$\text{mg biomass-C} \bullet 100\,\text{g}^{-1}\text{dm} = \frac{\text{A} - \text{B}}{0.35},$$

where A — mean value of organic C of samples (mg organic $\text{C}-100\,\text{g}^{-1}$ dm); B — mean value of organic C of controls (mg organic $\text{C}-100\,\text{g}^{-1}$ dm); and 0.35 — factor (k_{EC}) that converts the organic C-flush to microbial C.

The k_{EC}-values of different soils vary and are affected by soil moisture content. The optimum moisture content of soils lies at a water tension of -5 kPa.[47] The described method can thus also be used for soils with a low moisture content.

6.2. *Microbial Activity*

Soil microbial activity leads to the liberation of nutrients available for plants and to the mineralization and mobilization of pollutants and xenobiotics. Thus, microbial activity is of crucial importance in biogeochemical cycling.

The measurement of microbial parameters, such as soil respiration or enzyme activities, provides information on the presence and activity of viable microorganisms as well as on the intensity, kind and duration of the effects of waste disposal on soil metabolic activity. Such measurements may serve as a good index of the impact of pollution on soil health. Microbiological and biochemical properties are very responsive and provide precise information on small changes occurring in soil.[44,48–51]

Parameters such as microbial biomass C provide information on the size of these communities, while others are related to their general activity, such as respiration, ATP or dehydrogenase activity, or specifically related to the cycle of elements in the soil; such as the case with urease, phosphatase, and b-glucosidase hydrolases. The advantage of these parameters resides in their sensitivity to changes.[43,44,52–54] However, due to the soil biochemical and microbiological complexity, it is necessary to use several parameters when evaluating the soil's biochemical and microbial states.

6.3. *Soil Respiration*

Microbial respiration (soil respiration) is defined as oxygen uptake or carbon dioxide (CO_2) evolution by soil organisms.[55] Soil respiration results from the degradation of organic matter (e.g. mineralization of harvest residues). This soil biological activity consists of numerous individual activities; the formation of CO_2 is the last step of carbon mineralization. In undisturbed soils (no nutrient addition etc.), there will be an ecological balance between the organisms and their activities. The respiration is then called "basal respiration." Upon a disturbance, e.g. through addition of organic matter, one can observe a change in the soil respiration due to more rapid growth and a higher mineralization of the microorganisms. This increased respiration is characterized by an initial, an acceleration, an exponential, a delay, a stationary, and a decreasing phase. CO_2 evolution from a soil is thus a measure of the total soil biological activity.

Soil respiration can be determined in the field[42,56] as well as in the laboratory.[42,55,57,58] Field investigations are done with open measuring devices; however, the measurements are more labor-intensive and result in larger fluctuations than laboratory measurements. It is not possible to differentiate between the respiration from the microorganisms and from the vegetation cover and plant roots.

Laboratory investigations use soils homogenized by sieving under standardized conditions. A number of procedures can be used. More often the following methods are used. CO_2 measurements in a closed system: CO_2 from the soil accumulates in closed containers. The CO_2 can either be absorbed in alkaline solution, and then be quantified analytically or analyzed at the beginning and end of the incubation time with a gas chromatograph. The CO_2 accumulation can then be converted to hourly production rates.

6.3.1. *Soil Respiration by Titration*[59]

Soil samples are incubated in a closed vessel at 25°C. The CO_2 produced is absorbed in sodium hydroxide and quantified by titration.[46,55,60]

6.3.1.1. Procedure

Weigh 20–25 g of field moist soil into three centrifuge tubes or test tubes (samples). Pipette 20 ml of sodium hydroxide solution (0.05 M) into the laboratory bottles, an insert the tubes into the bottles. Close the bottles and incubate for 24 h at 25°C.

Remove the tubes and add 2 ml of barium chloride solution (0.5 M) to precipitate the adsorbed CO_2 as barium carbonate. Add 3–4 drops of indicator solution and titrate the remaining sodium hydroxide with diluted HCl (0.1 M).

To prepare indicator solution, dissolve 0.1 g of phenolphthalein in ethanol (60% v/v), and make up the volume to 100 ml with ethanol in a volumetric flask.

To prepare controls, perform procedure without soil. Use at least three samples and control replicates.

6.3.1.2. Calculation of results

$$\mathrm{mg\,CO_2 \cdot g^{-1} \cdot dm \cdot 24\,h^{-1}} = \frac{(\mathrm{C} - \mathrm{S}) \cdot 2.2 \cdot 100}{\mathrm{SW} \cdot \%\mathrm{dm}},$$

where C — mean volume of HCl consumed by controls (ml), S — mean volume of HCl consumed by samples (ml), 2.2 — conversion factor (1 ml of 0.1 M HCl corresponds to 2.2 mg CO_2), SW — initial soil weight (g), and $100 \cdot \%^{-}\mathrm{dm}^{-1}$ factor for soil dry matter.

6.4. ***Nitrogen Metabolism***

Bioavailable nitrogen is one of the keys for plant growth in agriculture. At the same time, nitrogen compounds such as nitrate, nitrite, or N_2O play an important role in environmental pollution. On the one hand, two main delivery processes (mineralization and nitrogen fixation) are known. On the other hand, nitrification and denitrification can cause significant losses of nitrogen from the bound pool.

6.4.1. *Nitrogen Fixation*

Nitrogen fixation is performed by phylogenetically and physiologically diverse groups of prokaryotic organisms. Fixation of atmospheric dinitrogen by biological enzyme systems can be performed by three main types of organisms in soils: free-living heterotrophic bacteria, phototropic cyanobacteria, and symbiotic associations.[25,33,61]

6.4.1.1. Nitrogenase activity by acetylene reduction[46,62]

Undisturbed soil cores, which can include plants or sieved soil samples, are incubated in gas-tight vessels in an atmosphere containing 10% volume of acetylene at 22°C for 1–48 h. During the incubation, gas samples are withdrawn several times to determine ethylene concentrations using a gas chromatograph.

6.4.1.1.1. *Procedure*

Fit the soil core (*ca.* 500 g of field-moist soil) into the Kilner jar and seal the vessel after spreading silicone grease onto the rubber rings. In the case of the sieved soil samples, weigh 7 g of soil into the serum vial and seal with rubber septum and aluminum cap.

For determination of potential nitrogenase activity, field-moist soil is amended with 1% glucose and mixed thoroughly (e.g. by shaking in an inflated plastic bag). The sample is then adjusted to field capacity by the addition of water and weighed into the serum vials. Incubate the vessels for up to 48 h at 22°C in a thermostatic incubator.

Five minutes after the addition of acetylene, inject a 500 μl gas sample into the gas chromatograph and analyze it for the initial ethylene concentration. According to the expected activity of the sample, repeat the measurement in 1–10 h intervals. Determine a calibration curve by injecting 100, 200, 300, 400, and 500 μl of ethylene standard into the gas chromatograph. After every 10 samples, inject a standard to control the instrument performance. Propylene can be used as internal standard to check for gas leaks in the vessels, whereas the use of propane is not appropriate.

Gas chromatography regime:

Column dimension: 2 m × 2 mm
Filling: Porapak R, 80–100 mesh (or Porapak N, T)
Detector: flame ionization detector (FID) with hydrogen/air flame
Carrier gas: nitrogen, flow rate: 20 ml • min^{-1}
Injector temperature: 70°C
Oven temperature: 50°C, isothermic
Detector temperature: 155°C
Mean retention times: ethylene: 1.16 min; acetylene: 1.87 min

6.4.1.1.2. *Calculation of results*

The calibration curve is used to calculate the concentration of ethylene (vpm = $\mu l \bullet l^{-1}$) in the headspace of the respective sample. Depending on the subject of investigation, the results can be expressed per soil weight, per soil volume, or per area.

After determination of a conversion factor with $^{15}N_2$, the results of the acetylene reduction assay can be converted into μg of fixed nitrogen. The theoretical conversion factor for soil samples is "4." However, practical experience has proved that it is necessary to determine this factor for each soil separately.

The nmol ethylene $\bullet g^{-1}$dm $\bullet h^{-1}$ is divided by the conversion factor and multiplied by 28 (molecular weight of N_2).

$$\text{nmol ethylene}\bullet \text{g}'\text{dm}\bullet \text{h}^{-1} = \frac{(\text{S} - \text{C}) \cdot \text{V} \cdot \text{P} \cdot 100}{\text{SW} \cdot \text{ml} \cdot \text{R} \cdot \text{T} \cdot \text{t} \cdot \%\text{dm}},$$

where S — amount of ethylene in injected sample after incubation (nl); C — amount of ethylene in injected sample before incubation (nl); V — headspace volume in assay vessel (ml); P — air pressure under standard conditions (101300 Pa); SW — initial soil weight (g); ml — amount of sample injected into the gas chromatograph (0.5 ml); R — gas constant ($8.314\ \text{J} \bullet \text{mol}''^{1} \bullet \text{K}''^{1}$); T — temperature of incubation (295 K); t — incubation time (h); and $100 \bullet \%\text{dm}^{-1}$ — factor for soil dry matter.

6.4.2. *Nitrification*

Nitrification is the chemolithoautotrophic oxidation of ammonium via nitrite to nitrate.[63–65] Nitrification can be measured directly using labeled nitrogen. Another possibility to determine potential nitrification is based on the addition of chlorate to inhibit the nitrite oxidation.[66] The potential nitrification can be measured as the accumulation of nitrite after the addition of ammonium in short-term experiments. This method is well suited for measuring potential nitrification in high number of samples. In some cases, incubation experiments were performed. Released nitrate is determined after an incubation period of some weeks under field or laboratory conditions.[63–65,67]

6.4.2.1. Potential nitrification[65]

Using ammonium sulfate as substrate, soil samples are incubated for 5 h at 25°C. Nitrite released during incubation period is extracted with potassium chloride and determined colorimetrically at 520 nm. Sodium chlorate inhibits nitrite oxidation during incubation.

6.4.2.1.1. *Procedure*

Weigh 5 g of field-moist soil into three 100-ml Erlenmeyer flasks. Add 20 ml of substrate working solution (ImM $(NH_4)_2SO_4$) and 0.1 ml of sodium chlorate solution (1.5 M), mix briefly, and close the tubes with caps.

The concentration of the inhibitor $NaClO_3$ has been optimized for arable soils with a humus content of 1.5–3.5%. If this method is used for the estimation of potential nitrification in grassland and forest soils, increase the concentration of the inhibitor in the soil suspension because of a possible nonspecific adsorption of $NaClO_3$ onto organic matter. Test inhibitor concentration in a preexperiment. Ammonium oxidation is inhibited by high amounts of ammonium. Test substrate concentration in a preexperiment. Incubate two tubes for 5 h on a rotatory shaker (samples) and store the third tube for 5 h at −20°C (control).

After incubation thaw the control at room temperature, add 5 ml of potassium chloride solution (2 M) to samples and control, mix briefly, and filter samples and control immediately. For photometric analysis pipette 5 ml of filtrates, 3 ml of ammonium chloride buffer (0.19 M pH 8.5) and 2 ml of color reagent into test tubes, mix, and allow to stand for 15 min at room temperature. Measure extinction of samples and control at 520 nm against the reagent blank. The color complex is stable for at least 4 h.

To prepare the color reagent dissolve 2 g of sulfanilamide and 0.1 g of *N*-(l-naphthyl)-ethy-lenediamine hydrochloride in 150 ml of distilled water and add 20 ml of cone, phosphoric acid. Cool the solution to room temperature and dilute the volume to 200 ml with distilled water in a volumetric flask. This solution has to be colorless and has to be prepared daily.

6.4.2.1.2. *Calculation of results*

Calculate μg N of the test solution from the calibration curve. Express potential nitrification as the amount of $NO_2{-}N$ released from 1 g of soil during 5 h.

$$\mathrm{ng\,N \bullet g^{-1}dm \bullet 5\,h^{-1}} = \frac{(\mathrm{S} - \mathrm{C}) \cdot 25.1 \cdot 1000 \cdot 100}{5 \cdot 5 \cdot \%\mathrm{dm}},$$

where S — mean value of samples (mgN), C — control (mgN), 25.1 — volume of extract (ml), 1,000 — conversion factor (1 mg N = 1,000 ng N), 5 — aliquot of filtrate (ml), 5 — initial soil weight (g), and $100 \bullet \%\mathrm{dm}^{-1-}$ — factor for soil dry matter.

6.4.2.2. Nitrification during long-term incubation[46,63]

Using ammonium chloride solution as substrate, soil samples are incubated for up to three weeks at 25°C. After extraction of inorganic nitrogen compounds with potassium chloride solution, the contents of ammonium and nitrate are determined colorimetrically. The oxidation of the substrate is equivalent to the nitrification dynamics and is expressed as percentage of the added nitrogen.

6.4.2.2.1. *Procedure*

Weigh 10 g of field-moist soil into four 100-ml Erlenmeyer flasks and add dropwise 1 ml of substrate solution (75.7 M $(NH_4)_2SO_4$). Adjust soil moisture to 50–60% of water-holding capacity with distilled water. Close the flasks with caps. Incubate two flasks up to three weeks at 25°C (samples) and store the third flask immediately at −20°C (control). Weigh flasks once a week to control soil moisture content. Adjust soil moisture content by adding distilled water, if necessary. After incubation, add 50 ml of potassium chloride solution (2 M) to both samples and control and shake the flasks for 30 min on a rotatory shaker. Filter flask contents and determine ammonium and nitrate.

For photometric analysis of ammonium pipette 5 ml of filtrate, 2.5 ml of reagent A, and 1 ml of sodium dichloroisocyanurate solution (39.1 mM) into a test tube, mix well, and allow to stand for 30 min at room temperature. Measure extinction of samples with a spectrophotometer at 600 nm against reagent blank.[68]

To prepare reagent A, mix 100 ml of 0.3 M NaOH solution and 100 ml of sodium salicylate solution with 100 ml of distilled water. Prepare immediately before use.

Calculate $\mu gNH_4^+{-}N$ in the test tube from the calibration curve.

$$\mu gN \bullet g^{-1}dm = \frac{S \cdot V \cdot F \cdot 100}{A \cdot SW \cdot \%dm},$$

where S — mean value of samples (μgN), V — volume of extraction mixture (ml), F — dilution factor, A — aliquot of filtrate (ml), SW — initial soil weight (g), and $100 \bullet \%dm^{-1}$ — factor for soil dry matter.

For analysis of nitrate, pipette 5–25 ml of clear filtrate into two test tubes, adjust to 25 ml with distilled water, add 1 ml of diluted sulfuric acid, and mix.

Add 3–4 granules of zinc to one of the test tubes and allow to stand overnight for the reduction of nitrate (control).

Measure the extinction of samples and controls with a spectrophotometer at 210 nm against the reagent blank.

Calculate $\mu gNO_3{-}N$ in the test tubes from the calibration curve.

$$\mu g\,N \bullet g^{-1}dm = \frac{(S - C) \cdot V \cdots 100}{A \cdot SW \cdot \%dm},$$

where S — sample (filtrate without zinc; μgN), C — control (filtrate with zinc; μgN), V — volume of extract (ml), A — aliquot of filtrate (ml), SW — initial soil weight (g), and $100 \bullet \%dm^{-1}$ — factor for soil dry matter.

6.4.2.2.2. *Calculation of results*

Take into consideration that inorganic nitrogen can be released from organic nitrogen (ammonification), and that both ammonium and nitrate can be immobilized by microorganisms during incubation.

$$(\mathrm{NO_3^-} - \mathrm{N} + \mathrm{NH_4^+} - \mathrm{N})_\mathrm{A} - (\mathrm{S} + \mathrm{NO_3^-} - \mathrm{N} + \mathrm{NH_4} - \mathrm{N})_\mathrm{B} = \mathrm{NA},$$

where S — amount of initially added $\mathrm{NH_4^+}$–N (t_0; μg N • $\mathrm{g^{-1}}$dm); A — inorganic nitrogen after incubation (sum of $\mathrm{NH_4^+}$–N and $\mathrm{NO_3^-}$–N; μgN–$\mathrm{g^{-1}}$dm); B — initial inorganic nitrogen (t_0; μgN–$\mathrm{g^{-1}}$dm); NA — correction factor for ammonification and immobilization during incubation time; $\mathrm{NA} < 0$: higher ammonification than immobilization; and $\mathrm{NA} > 0$: higher immobilization than ammonification.

6.4.2.2.3. *Nitrification turnover*

The nitrification turnover is equivalent to the amount of nitrogen that is released from the substrate per day and gram dry matter. Express results as percentage loss of the initially added substrate

$$\%\mathrm{N} \bullet \mathrm{d^{-1}} = \frac{(\mathrm{NO_3^-} - \mathrm{N_A} - \mathrm{NO_3^-} - \mathrm{N_B}) \cdot 100}{(\mathrm{S} + \mathrm{NA})\mathrm{n}},$$

where A — $\mathrm{NO_3^-}$–N content after incubation (μgN • $\mathrm{g^{-1}}$dm); B — initial $\mathrm{NO_{3-}}$–N content (t_0; μgN • $\mathrm{g^{-1}}$dm); S — amount of initially added $\mathrm{NH_4^+}$–N (t_0; μgN • $\mathrm{g^{-1}}$dm); NA — correction factor for ammonification and immobilization during incubation; *n* — incubation time (*d*); and %N • $\mathrm{d^{-1}}$ — nitrification turnover.

6.5. *Enzyme Activities*

Soil enzyme activities can be used as an index of microbial functional diversity.[21,48,69,70,71] Since enzymatic activities in soil are mainly of microbial and fungal origins, the characterization of soil enzyme pattern could supply us information on microflora activity, soil productivity, and impact of pollutants.[72] Soil enzyme activities are considered as sensitive and early indicators of both natural and anthropogenic disturbances.[6,31] There is also growing interest in using soil enzymes as indicators of soil fertility, since enzyme activities are sensitive to numerous factors such as climate, type of amendment, agricultural management practices, crop type, edaphic properties, and wastes disposal.[73,74]

Dehydrogenase is considered as good indicators of microbial activity in soil in relation to their mineralizing function.[52,75] Enzymatic activities in relation to the cycle of N (ammonification, nitrification, and denitrification) or P (release of inorganic P) in soil have been used to evaluate the fertility of the soil or to describe the functioning of the ecosystem.[76]

Nitrogen is transformed in the cycle as ammonium. Extracellular proteases are produced by microbes and secreted to the environment to hydrolyze macromolecular polypeptides into smaller molecules, which can be removed by the cell. Upon the release of proteases from microbial cells, these soil enzymes become physically adsorbed onto soil colloids or covalently bound to soil organic matter. These immobilized enzymes show a high degree of resistance to proteolysis. Numerous methods have been used for the assay of protease activity in soils. Most of these methods differ in the used substrate, incubation time, incubation temperature, and the estimation of the released product. When gelatin is used as substrate, an incubation time of at least 16 h is necessary. Several methods use casein as substrate to reduce incubation time to 2 h.[77,78]

The rapidly increasing importance of urea in agriculture has drawn attention to the need for methods to estimate its hydrolysis in soil. A number of different methods have been developed to estimate urea hydrolysis. Most methods involve the determination of the amount of ammonium released after incubation of soil with aqueous or buffered urea solution[79,80] whereas some methods measure the loss of urea.[81]

6.5.1. *Dehydrogenase Activity*

6.5.1.1. Dehydrogenase activity with the substrate triphenyltetrazolium chloride (TTC)

Soil samples are suspended in a TTC solution and incubated for 16 h at 25°C. The triphenyl formazan (TPF) produced is extracted with acetone and measured photometrically at 546 nm,[82] modified.

6.5.1.1.1. *Procedure*

Weigh 5 g of field-moist soil into four test tubes. Add 5 ml of substrate solution (0.1–2% 2,3,5-TTC to three tubes (samples). Pipette 5 ml of Tris buffer (0.1 M) to the fourth tube (control). Mix, close the test tubes with rubber stoppers, and incubate for 16 h at 25°C.

To extract the produced TPF, add 25 ml of acetone to both samples, and control. Mix and shake the tubes for 2 h in the dark.

Subsequently, filter the solutions in a semi-dark room. Measure the extinction of the filtrates and calibration standards photometrically at 546 nm within 1 h.

6.5.1.1.2. *Calculation of results*

Determine the μg TPF in the filtrates from the calibration curve.

$$\mu\text{gTPF} \cdot \text{g}^{-1}\text{dm} \cdot 16\,\text{h}^{-1} = \frac{(\text{S} - \text{C}) \cdot 100}{5 \cdot \%\text{dm}},$$

S — mean value of samples (μgTPF), C — control (μgTPF), 5 — initial soil weight (g), and 100 factor for soil$\cdot \%^{-1}$dm — factor for soil dry matter.

6.5.1.2. Dehydrogenase activity with the substrate INT[46,83]

Soil samples are mixed with (2-(*p*-iodophenyl)-3-(*p*-nitrophenyl)-5-phenyl tetrazoliumchloride) solution (INT), and incubated for 2 h at 40°C. The reduced iodonitrotetrazolium formazan (INTF) is extracted with dimethylformamide and ethanol and measured photometrically at 464 nm.

6.5.1.2.1. *Procedure*

Weigh 1 g of field-moist soil into three test tubes and add 1.5 ml of Tris buffer (1 M pH 7) and 2 ml of substrate solution. Stopper the test tubes, shake briefly, and incubate for 2 h at 40°C in the dark (samples). Prepare the control with autoclaved soil (autoclave for 20 min at 121°C and 1.1 bar), and proceed as with the samples. After incubation, add 10 ml of the extraction solution to each of the tubes. For extraction of the developed INTF, keep the tubes for 1 h at room temperature in the dark and shake vigorously every 20 min. Immediately after filtering the flask contents, measure the INTF of samples, control and calibration standards photometrically at 464 nm against the reagent blank.

To prepare substrate solution mix 500 mg of 2-(*p*-iodophenyl)-3-(*p*-nitrophenyl)-5-phenyl tetrazoliumchloride (INT, Serva 26840) with 2 ml of *N*,*N*-dimethylformamide and shake vigorously. Make up to volume with distilled water in a 100-ml volumetric flask, and dissolve in an ultrasonic bath. Prepare this reagent daily and store it in the dark until use.

To prepare the extraction solution mix 100 ml of *N*,*N*-dimethylformamide with 100 ml of ethanol (96% v/v).

6.5.1.2.2. *Calculation of results*

Dehydrogenase activity is expressed as μg of iodonitrotetrazolium formazan (INTF) per gram dry matter and incubation time.

$$\mu\text{gINTF} \cdot \text{g}^{-1}\text{dm} \cdot 2\,\text{h}^{-1} = \frac{(\text{S} - \text{C}) \cdot 100}{\%\text{dm}},$$

where S — mean value of samples (μgINTF), C — control (μgINTF), and $100 \bullet \%^{-1}$dm — factor for soil dry matter.

6.5.2. *Protease Activity*[46,84]

Using casein as substrate, soil samples are incubated for 2 h at 50°C and pH 8.1.

Amino acids released during the incubation period are extracted and the remaining substrate is precipitated after the addition of trichloroacetic acid. Aromatic amino acids react with Folin-Ciocalteu's phenol reagent in an alkaline solution to form a blue complex that is determined colorimetrically.

6.5.2.1. Procedure

Weigh 1 g of field-moist soil into four 25-ml Erlenmeyer flasks. Add 5 ml of substrate solution (2% casein sodium salt), and 5 ml of Tris buffer (0.05 M, pH 8.1) to two flasks (samples), and pipette only 5 ml of Tris buffer into the residual flasks (controls).

Shake the flasks briefly, seal with rubber stoppers, and incubate for 2 h at 50°C on a rotatory shaker. After incubation, pipette 5 ml of substrate solution to each of the controls. Shake briefly and add 5 ml of trichloroacetic acid solution (0.92 M) to both samples and controls. Shake briefly, filter samples, and controls immediately. For photometric analysis, pipette 5 ml of filtrate and 7.5 ml of alkali reagent into a test tube, mix well, add 5 ml of Folin-Ciocalteu's phenol reagent, and mix again.

To prepare the alkali reagent, mix 1,000 ml of reagent A, 20 ml of reagent B, and 20 ml of reagent C in a beaker. Prepare this working solution daily.

6.5.2.1.1. *Reagent A*

Weigh 50 g of sodium carbonate (water-free) in a 1,000-ml volumetric flask, add 60 ml of 0.1 M NaOH, dissolve in 600 ml of distilled water, and make up to volume with distilled water. Store the solution at 4°C no longer than three weeks.

6.5.2.1.2. *Reagent B*

Weigh 5 g of copper(II)sulfate pentahydrate in a 1,000-ml volumetric flask, dissolve in distilled water, and dilute to volume with distilled water. Store the solution at 4°C up to three weeks.

6.5.2.1.3. *Reagent C*

Weigh 10 g of sodium potassium tartate ($C_4H_4KNaO_6{-}4H_2O$) in a 1,000-ml volumetric flask, dissolve in distilled water, and make up to volume with distilled water. Store the solution at 4°C no longer than some days.

6.5.2.1.4. *Folin-Ciocalteu's phenol reagent*

Pipette 20 ml of Folin-Ciocalteu's phenol reagent (Merck 9001 or Sigma F–9252) in a 200-ml volumetric flask and make up to volume with distilled water.

If a precipitate is formed after 1 h, filter or centrifuge solutions again. Filter calibration standards and follow the same procedure as described for filtrates of samples and controls.

Allow to stand for exactly 90 min at room temperature for color development. Within the following 90 min, measure the extinction of calibration standards, samples and controls at 700 nm with a spectrophotometer against the reagent blank.

6.5.2.2. Calculation of results

Protease activity is expressed as μg tyrosine equivalents (tyr) per gram dry matter and incubation time. Tyrosine equivalents are calculated from the calibration curve.

$$\mu\text{g tyr} \bullet \text{g dm}^{-1} \bullet 2\,\text{h}^{-1} = \frac{(\text{S} - \text{C}) \cdot 100}{\%\text{dm}},$$

where S — mean value of samples (μg tyr), C — mean value of controls (μg g), and 100• %–′dm — factor for soil.

6.5.3. *Urease Activity*[46,67]

After the addition of an aqueous or a buffered urea solution, soil samples are incubated for 2 h at 37°C. Released ammonium is extracted with potassium chloride solution and determined by a modified Berthelot reaction. The determination is based on the reaction of sodium salicylate with NH_3 in the presence of sodium dichloroisocyanurate that forms a green-colored complex under alkaline pH conditions. Sodium nitroprusside is used as a catalyst and increases the sensitivity of the method about tenfold.

6.5.3.1. Unbuffered method

Weigh 5 g of field-moist soil into three 100-ml Erlenmeyer flasks. Add dropwise 2.5 ml of substrate solution (79.9 mM urea) for the unbuffered method to two flasks (samples) and pipette 2.5 ml of distilled water into the residual flask (control). Stopper the flasks and incubate for 2 h at 37°C. After incubation, add 2.5 ml of substrate solution for the unbuffered method (79.9 mM urea) to controls and 2.5 ml of distilled water to samples. Add 50 ml of 2 M potassium chloride solution to both samples and controls and shake for 30 min on a rotatory shaker. Filter the contents of the flasks through folded niters.

For photometric analysis, pipette 1 ml of filtrate, 9 ml of distilled water, 5 ml of reagent A, and 2 ml of 39.1 mM sodium dichlorisocyanurate solution (39.1 mM) into a test tube, mix well, and allow to stand for 30 min at room temperature.

To prepare reagent A mix 100 ml of 0.3 M NaOH solution and 100 ml of sodium salicylate solution (1.06 M) with 100 ml of distilled water. Prepare this reagent immediately before use.

Calculate μgN $\bullet$ ml^{-1} from the calibration curve.

6.5.3.1.1. *Calculation of results*

$$\mu\text{gN}\bullet\ \text{ml}^{-1} = \frac{(\text{S} - \text{C}) \cdot 10 \cdot 55 \cdot 100}{5 \cdot \%\text{dm}},$$

where S — mean value of samples (μg $\bullet$ ml^{-1}), C — control (μg $\bullet$ ml^{-1}), 10 — dilution factor, 55 — volume of extract (ml), 5 — initial soil weight (g), and 100 $\bullet$ % dm — factor for soil dry matter.

6.5.3.2. Buffered method

Weigh 5 g of field-moist soil into three 100-ml Erlenmeyer flasks. Add 2.5 ml of substrate solution (720 mM urea) for the buffered method and 20 ml of 0.1 M, pH 10 borate buffer to two flasks (samples), pipette only 20 ml of borate buffer into the residual flask (control). Shake the flasks briefly, seal with rubber stoppers, and incubate for 2 h at 37°C.

After incubation, pipette 2.5 ml of substrate solution (for the buffered method) to the control, add 30 ml of 2 M potassium chloride solution to both samples and control, and shake for 30 min on a rotatory shaker. Filter the contents of the flasks through folded niters.

For photometric analysis, follow the procedure for the unbuffered method.

Calculate μgN $\cdot$ ml^{-1} from the calibration curve.

6.5.3.2.1. *Calculation of results*

$$\mu\text{gN} \cdot \text{g}^{-1} \cdot \text{dm} \cdot 2\,\text{h}^{-1} = \frac{(\text{S} - \text{C}) \cdot 10 \cdot 52.5 \cdot 100}{5 \cdot \%\text{dm}},$$

where S — mean value of samples (μgN$\bullet$ml^{-1}), C — control (ug N- ml^{-1}), 10 — dilution factor, 52.5 — volume of extract (ml), 5 — initial soil weight (g), and 100$'$ $\cdot$ %$^{-1}$dm — factor for soil dry matter.

7. Bioassays

7.1. *Toxicity Identification Evaluation (TIE)*

Toxicity identification and evaluation procedure, developed by US Environmental Protection Agency,[65,85] makes possible to determine what fraction of chemicals causes the observed impact on the bioassays. TIE methods have proven useful in identifying the toxicity of organic contaminants, ammonia, and metal in soil leachates, effluents, sediment pore waters, and leachates from products such as car tires. These methods allow to fractionate the toxicity of different metals.[86] Daphnids are the most commonly used organism in TIE. For solid samples, the seeds of plants are relevant.

For solid samples, the following TIE procedure is recommended. At the beginning, immobility test with *Dapnia magna* and root elongation test with *Latuca sativa* with soil or waste extracts are carried out. After the initial toxicity test with *D. magna*, the samples elutriates are diluted to their 48 h EC_{48s} with reconstituted standard water, or, when testing the effect of pH manipulations, with soft water. This concentration is since most TIE treatments have a reducing effect on toxicity.

After the initial toxicity test with lettuce (*L. sativa*), the waste or soil extracts are diluted to their 96 h EC_{50s} with pure water. These samples, with and expected decrease in root elongation of 50%, are then poured into Petri dishes after TIE treatments as described below.

Eight TIE treatments are carried out:

(1) EDTA addition to a final concentration of 40μM, which reduces toxicity of metal ions,
(2) sodiumthiosulfate (STS) addition to a final concentration of 60μM, which reduces toxicity of silver and oxidizing compounds,
(3) filtering through a CM-resin, which reduces the toxicity of cations,
(4) filtering through a C18 resin, which reduces toxicity of lipophilic compounds,
(5) filtering through a QMA-resin, which reduces the toxicity of anions,
(6) filtering through a Millex resin 0, 2μM, which reduces toxicity of large complexes, and
(7 and 8) adjustment of pH by decreasing it to 6 or increasing it to 9 (Table 4).

The sample in the first well or Petri dish is left untreated for comparison, and the effect in this untreated sample should be approximately 85% immobility of *D. magna* and 50% decreased root elongation for lettuce. After treatment, the samples are left for equilibration for an hour before ten newborn *D. magna* or ten lettuce seeds are added to each well. The immobility of the daphnids and root length of the lettuce seeds, in the differently treated samples, are determined after 48 h and 96 h, respectively, and compared to the untreated samples. The number of replicates is three. After the TIE treatment, the effects of different parameters (metals, complex compounds, etc.) are analyzed.[87]

Table 4. TIE Treatments for *D. magna* and *L. sativa* and Their known Effects (− or +) on Different Compounds.[87]

	Addition of		Filtration through				Adjustment to	
TIE Treatment	EDTA (40μM)	STS (60μM)	CM-resin	C18-resin	QMA-resin	Millex-resin (0, 2 μm)	pH 6	pH 9
Effects on toxicity	Metal−	Ag−	Metal−	Lipophilic compounds−	Anions−	Large complexes−	Metals+	Metals−
		Oxidizing compounds−	Ammonia−				Acid+	Acid−
							Bases−	Bases+

7.2. *Elutriate Bioassays*

7.2.1. *Solid Samples Extraction*

For the soil and waste extraction, 5 g of the sample were stirred for 1 h in demineralized water at a ratio of 1:10 (weight/volume) on a shaker at 200 rpm, left untouched overnight, and then centrifuged at 4,300 g for 10 min. For all tests, the starting water extract and its dilution were examined. If the 100% inhibition of the starting extract was revealed, the series of the dilution was then prepared and examined. The number of the dilution depended on the toxicity of the starting water extract.

7.2.2. *RET Assay*

The RET assay is an enzymatic *in vitro* screening test based on the effects of toxic compounds on NAD reduction by submitochondrial particles. The reaction conditions are arranged so that the enzyme NADH-coenzyme Q reductase is operating in reverse direction. Prepare submitochondrial particles from beef heart mitochondria. Before the test, dilute sample extracts with a concentrated assay buffer to achieve final concentrations of 50%. The reaction is followed spectrophotometrically at 340 nm for *ca.* 10 min. Use a linear portion of the reaction is used to calculate the increase in absorbance units per minute. The inhibition of the reaction is calculated by comparing the reaction rate of the samples to the controls. Deionized water or DMSO-water is used for the controls.

7.2.3. Ceriodaphnia dubia *Reproduction Test*

Bioassay is performed according to the AFNOR (French Association of Standardization) guidelines for conducting chronic toxicity tests with *C. dubia.*

Introduce individually young organisms less than 24 h old and from the second to the fifth brood of healthy adults in 100-ml glass vials containing 50 ml of test solution (Evian mineral water with Ca = 78 mg/l, Mg = 24 mg/l, Na = 5 mg/l, K = 1 mg/l, SO_4 = 10 mg/l, Cl = 4.5 mg/l). Measure hardness and pH of the test solutions are daily. The recorded values always must satisfy the criteria of acceptance of the AFNOR guidelines: set to 8 ± 0.3 for the pH and 250 ± 20 mg/l as $CaCO_3$ for the hardness. All the tests have to be conducted at 25 ± 2°C. Temperature must be continuously measured during the experiments. A 16:8 light:dark photoperiod must be employed. Light intensity, measured at the beginning of each test, at the air/water interface must range between 300 and 500 lx. Dissolved oxygen must range between 8.6 and 9.2 mg/l. Change the solutions at days: $d + 3$, $d + 4$,

$d + 5$, and $d + 6$. Feed *C. dubia* with *Chlorella vulgaris* ($6 * 10^6$ cells/50 ml), *Raphidocelis subcapitata* ($3 * 10^6$ cells/50 ml), and 25μl/50 ml of a solution at 5 g/l of SERAMICRON®, a commercial fish food. Use 10 replicates, with one individual in each, per treatment and for the control. Choose a range of five elutriate dilutions those causing 0–100% of effect on the reproduction of *C. dubia* for each tested sample. The criterion for estimating the effects of the wastes on the reproduction of *C. dubia* is the effective concentration inducing an inhibition of 10% of the reproduction of the tested organisms (EC_{10}). Acceptability of all test results requires the following conditions: (1) control survival of greater than 80%, (2) less than 20% of males at day 7 in each batch, (3) more than 60% of control females have had three broods, and (4) control females producing an average of at least 15 offspring in day 7. In addition, the results of routine reference toxicant tests on sodium pentachlorophenate (C_6Cl_5ONa) and copper(II)sulfate pentahydrate ($CuSO_4^*5H_2O$) have to fall within historical ranges of acceptability (AFNOR), indicating that the organisms are healthy.[88]

Calculate the inhibition as following:

$$I(c) = 100 - \frac{\text{Number of } C.\ dubia \text{ survived in tested sample} \cdot 100}{\text{Number of } C.\ dubia \text{ survived in control sample}},$$

where I (c) — Inhibition of the tested sample or of the chosen waste dilution.

7.2.4. *Spirotox Test*

The Spirotox test utilizes a large ciliate protozoan *Spirostomum ambiguum* as a test organism. *S. ambiguum* is one of the biggest protozoans at 2–3 mm long. Two different test functions can be measured in the study: lethality and deformation of *S. ambiguum*. Often, deformation is more sensitive than lethality.

The Spirotox test is based on the Spirotox-volatile procedure. Use disposable 24-well microplates as test containers. Use Tyrod solution (1:64) as a diluent. It is made up from 125 mg NaCl, 3.1 mg KCl, 3.1 mg $CaCl_2$, 1.55 mg $MgCl_2$, 15.6 mg $NaHCO_3$, and 0.78 mg NaH_2PO_4 per L of deionized water. Total hardness has to be about 2.8 mg $CaCO_3$/L and pH about 7.4 ± 0.2. Prepare a five-step dilution series in triplicate directly in the microplate. Grease the ring of each well with silicone fat after introduction of the protozoa into the wells. Then, cover the microplate with a sheet of polyethylene film and tightly close with the lid.

After incubation in the dark at 25°C, observe two kinds of test responses after 24 and 48 h of incubation: (1) different deformations, which mean morphological changes such as shortening, bending of the cell, etc. and (2) lethal response — spherical deformation and autolysis. Calculate two values — EC_{50} and LC_{50} — by graphical interpolation of test response versus toxicant concentration (log scale). The graphical method does not provide interval for the $LC(EC)_{50}$ estimate. This method is chosen because there were no partial toxic effects. Each chemical has to be assayed for a minimum of three replicates. Stock dilutions of toxicants are prepared in deionized water or, if a toxicant is insoluble in water, in methanol. When methanol is used as solubilizer, its concentration in test samples does not exceed 1%. The same concentration of methanol in Tyrod has to be used as a control.[89]

7.2.5. Pseudomonas putida *Growth Inhibition test*[90]

The bacterium *P. putida* represents a common aquatic heterotrophic microorganism. When *P. putida* cells are cultured under specified conditions, in a defend medium with different concentrations of samples over several generations, toxic substances present in the sample can inhibit the cell multiplication of the bacteria.

For the toxicity testing with *P. putida*, add 10 ml of the inoculum, which has been adjusted to a specific turbidity ($A_{610} = 0.1$), to the culture flask filled with 80 ml of the cultural medium and 10 ml of the test sample. The cultural medium contains ($\mathrm{mg\,l^{-1}}$): $NaNO_3$–500, K_2HPO_4–120, KH_2PO_4–60, glucose–2000, $MgSO_4$–200, iron(III)citrate–0.5. After an incubation period of 16 h ± 1 h at a constant temperature of 23°C in the dark, measure turbidity after homogenization by shaking. The number of replicates test is three. The percentage of inhibition is determined by comparing the response given by a test solution to the control solution. An inhibition curve is fitted to calculate LID_{10}

$$\text{Inhibition,\%} = 100 - \frac{\text{Extinction of the test solution} \cdot 100}{\text{Extinction of the control sample}}.$$

7.2.6. *BioTox*[91]

The inhibition of light production by *Vibrio fischeri* indicates disturbance of the energy metabolism of this heterotrophic bacterium. The luminescence pathway is a direct branch of the electron transport chain, and from the luminescence measurement the metabolic status of this bacterium can be. The change in bacterial luminescence when these bacteria are exposed to waste or soil extract samples can be used as an indicator of potential toxicity. Tests can be performed with soil water elutriates or with fresh homogenized soil samples with solid-phase modification of the standard test.[92]

The luminescence inhibition test is accomplished by combining 500μl elutriate with 500μl luminescent *V. fischeri* suspension. The test tubes are then incubated in a 15 ± 1°C water bath. Luminescence is measured after 30-min incubation with a luminometer. The inhibition of luminescence is calculated by comparing sample luminescence with that of the control containing deionized water

$$\text{Inhibition,\%} = 100 - \frac{\text{Luminescence of the test solution} \cdot 100}{\text{Luminescence of the control sample}}.$$

7.2.7. Paramecium caudatum *Immobilization Test*[8]

The toxicity tests with *P. caudatum* are carried out in a microplate with holes containing 0.3 ml of the test solution (soil or waste extracts and their dilutions) and examined under a microscope. Ten individuals are exposed per hole for 1 h. After 1 h, mortality is determined. The number of replicates for each dilution and control (distilled water) is three. The percentage of inhibition (I,%) for each of the dilutions is determined by comparing the number of immobilized or died test organisms to the starting number for the organisms. Fit inhibition curve to calculate LID_{10}

$$\text{Inhibition, \%} = 100 - \frac{(\text{Ns} - \text{Nim}) \cdot 100}{(\text{Nsc} - \text{Nimc})}$$

where Ns — average starting number of *P. caudatum* in the sample (dilution); Nim — average number of immobilized or died *P. caudatum* in the sample (dilution); Nsc — average starting number of *P. caudatum* in the control sample; and Nim — average number of immobilized or died *P. caudatum* in the control sample.

7.2.8. Daphnia magna *Immobilization Test*

The tests with *D. magna* are performed in 50 ml beakers, filled with 20 ml of the test solution where five test organisms (aged 6–24 h) are subsequently added to each solution. Daphnids are not fed during the experiment. After 24 and 48 h, the number of immobilized specimens (mortality) is determined visually. The number of replicates test is three. Each test is accompanied by a positive control test series with the reference compound potassium dichromate. The 24-h EC_{50} of potassium dichromate is approximately 1.1 mg/l. No immobility has to be observed in the negative control with dilution water. The percentage of inhibition (I,%) for each of the dilutions is determined by comparing the number of immobilized or died test organisms to the starting number for the organisms

$$\text{Inhibition, \%} = 100 - \frac{(\text{Ns} - \text{Nim}) \cdot 100}{(\text{Nsc} - \text{Nimc})}$$

where Ns — average starting number of *D. magna* in the sample (dilution); Nim — average number of immobilized or died *D. magna* in the sample (dilution); Nsc — average starting number of *D. magna* in the negative control; and Nimc — average number of immobilized or died *D. magna* in the negative control.

An inhibition curve has to be fitted to calculate LID_{10}.[93]

7.2.9. *Terrestrial Plants*

Plants arguably constitute the most important component of the world's ecosystem because of their unique ability to capture solar energy and transform it into chemical energy, while absorbing carbon dioxide and releasing oxygen into the environment. The sensitivity of plants to chemicals varies considerably. Plants sensitive to harmful substances can be used as test objects. Among the plant tests, phytotoxicity tests (toxicity and hazard posed by pollutants to the growth and survival of plants) have received the most attention. Phytotoxicity can be determined as seed germination, root elongation, and seedling growth.[12]

7.2.9.1. Raphanus sativus *root elongation test*

R. sativus is one of the sensitive plants that is used in elutriate phytotoxicity tests. The experiments are carried out on filter paper in Petri dishes. Put 5 ml test solution (or waste dilution) into the dish (number of replicates is three). Demineralized water serves as a control. Then, place 25 radish seeds (*R. sativus*) on the filter paper and put the dishes in a germination chamber at 20°C. After three days, measure the root lengths. The percentage of inhibition is determined by comparing the response given by a test solution to the control solution.[94]

$$\text{Inhibition, \%} = 100 - \frac{\text{Ls} \cdot 100}{\text{Lc}}$$

Ls — average root length of *R. sativus* in the tested sample (dilution)
Lc — average root length of *R. sativus* in the control.

7.2.9.2. Lemna minor *growth inhibition test*[95]

In the beginning of the test, 10 young plants with two leaves (fronds) are transferred into vessels containing 15 ml of the test solution. The soils or waste extracts are mixed with a buffered-nutrient medium in 50% and 10% final concentrations. The vessels are placed under a light source (4000–5,000 lx, light dark period of 16/8). The temperature is held at 20 ± 2°C and the relative humidity at 40–60%. Three replicates of the controls, the tested solutions, and the positive controls with potassium dichomate (5 mg Cr/liter) are tested. The number of fronds is calculated on day 3 and at the end of the test on day 5. Fresh weight and chlorophyll-a content are also measured. Chlorophyll is extracted by homogenizing the plant tissue in ethanol. The homogenate is stored at 20°C for at least 24 h and centrifuged to remove the tissue. The clear supernatant is used for spectrophotometric determination of chlorophyll-a. Inhibition of *L. minor* growth is calculated using number of fronds, fresh weight, and chlorophyll-a concentrations.

$$\text{Inhibition average, \%} = \frac{I_{\text{fronds}} + I_{\text{chlorophyll}} + I_{\text{biomass}}}{3}$$

$$I_{\text{chlorophyll}}, \% = 100 - \frac{\text{Es} \cdot 100}{\text{Ec}}$$

$$I_{\text{fronds}}, \% = 100 - \frac{\text{Ns} \cdot 100}{\text{Nc}}$$

$$I_{\text{biomass}}, \% = 100 - \frac{\text{Ws} \cdot 100}{\text{Wc}},$$

where Es — average optical density of the sample (dilution), Ec — average optical density of the negative control, Ns — average number of fronds by presence of the sample (dilution), Nc — average number of fronds by presence of the negative control, Ws — average fresh weigh of plants by presence of the sample, and Wc — average fresh weigh of plants by presence of the negative control.

7.2.9.3. Latuca sativa root elongation test

The dilution series, performed in three replicates for each waste or soil extract, for toxicity tests with lettuce seeds (*L. sativa*) are conducted in pure water. The concentrations of the dilution series are chosen depending on pollution range. Three replicates with 50 ml of each sample are poured into Petri dishes. Ten lettuce seeds are added to each Petri dish. The seeds are incubated at room temperature (22 ± 2°C) in darkness for 96 h. The root lengths are then measured. As negative controls, three Petri dishes with deionized water are used.[96]

The percentage of inhibition is determined by comparing the response given by a test solution to the control solution.

$$\text{Inhibition,\%} = 100 - \frac{\text{Ls} \cdot 100}{\text{Lc}},$$

where Ls — average root length of the plant in the tested sample (dilution) and Lc — average root length of the plant in the control.

7.2.10. Salmonella *Mutagenicity of Soils*

The *Salmonella* mutagenicity test is undoubtedly the most popular bioassay in environmental mutagenesis research, particularly for the analysis of complex mixtures such as organic extracts of soil and waste. The standard version of the assay, known as the plate incorporation assay, is a reverse mutation test that quantifies the frequency of reversion from histidine auxotroph to wild-type following a 48–72-h incubation with the test substances. Several tester strains of *Salmonella* are available, carrying a variety of his mutations. The most popular tester strains, TA98 and TA100, carry the hisD3052 and hisG46 alleles, respectively. The mutagenic potency is expressed as net revertents per unit of extractable organic material (e.g., mg extractable residue or milliliter of solvent extract). Different extract medium are used in different modifications of the method: MetOH; hexane/acetone; DCM; cyclohexane, or solvent mixtures (e.g., hexane/isopropanol, DCM/MetOH, benzene/EtOH, etc.).[10]

7.3. *Contact Bioassays*

7.3.1. *Terrestrial Plant Root Elongation and Biomass Tests*

As in elutriate plant bioassays, in contact bioassay, phytotoxicity parameters such as inhibition of seed germination, effects on root elongation, or plant growth are mostly measured. Photosynthesis, respiration, and enzyme activities, as well as tissue cultures, are not used as standard tests. The tests are carried out in pots with 50–100 g of tested samples. Ten seeds of the test plant (e.g. ryegrass (*Lolium perenne*), rice (*Oryza sativa*), wheat (*Triticum aestivum*), rape (*Brassica napus*), radish (*R. sativus*), fenugreek (*Trifolium ornithopoides*), cabbage (*B. oleracea*), corn (*Zea mays*), onion (*Allium cepa*), soybean (*Glycine max*) etc.) are put into the moistened solid sample and incubated for two weeks by 25 ± 2°C, 16:8 light:dark photoperiod and appropriate watering. The number of replicates is three. Uncontaminated soil is used as a control. After the incubation, the chosen phytotoxicity parameter (e.g. root length or biomass (fresh weight) of the plant) is measured and compared with control.

7.3.1.1. Trifolium pratense *germination test*[9]

For the red clover (*T. pratense*) seed germination test, layer 30 g of solid sample on plastic Petri dish (diameter 14 cm) and moisten with 20 ml of deionized water. Use two replicates per sample and for the control. Cover the dishes and incubate at room temperature for 4 h before the seeding. Divide the area of the dishes by plastic strips into four sectors. Place visibly healthy red clover seeds on the soil substrate, 20 seeds in every sector. Use artificial soil as the reference soil substrate (control). The test vessels are incubated in the dark at 25°C and the germinated seeds are counted on days 4 and 7. Seeds having a primary root 5 mm or longer are defined as germinated. The percentages of germinated seeds are calculated and compared with those of controls.

$$\mathrm{I}, \% = 100 - \frac{\mathrm{Ns} \cdot 100}{\mathrm{Nc}},$$

where Ns — number of germinated seeds in the sample and Nc — number of germinated seeds in the control.

7.3.2. Allium cepa *anaphase Aberration Test*

Allium spp. have large chromosomes ($2n = 16$) that are well suited to the scoring of chromosome aberrations, and tests for studying the genetic effect of chemicals on *Allium* chromosomes date back as far as 1938.

Onion bulbs or seeds are germinated and 1–2-cm-long roots are then exposed to previously moistened soils, soil slurries, or soil leachates, etc. for 2–24 h (usually one mitotic cycle). The number of replication is three. As a control, the artificial soil is used. Roots are then fixed, stained and 100 anaphase or telophase cells scored for aberrations including chromosome fragments and bridges.

Some researchers also score the frequency of vagrant chromosomes and multipolar cells that are presumed to be the result of c-mitotic events.[10]

7.3.3. *Bioluminescent Direct Contact Test*

For the solid-phase luminescent bacterial test, homogenized soil or waste samples of 5 g are weighed to 50-ml centrifuge tubes and diluted in 16 ml deionized water. This suspension is amended with 2 ml of 20% NaCl solution and with an inoculum of 2 ml of overnight-grown *V. fischeri* luminescent bacteria. Tubes are incubated for 15 min at 15 ± 1°C in a water bath. After incubation, the tubes are quickly centrifuged (5 min, 1660 rpm, 300 g) to separate solids from the exposed bacteria. The luminescence of a 1-ml sample of this supernatant is measured with a luminometer. The sample luminescence is then compared with luminescence in deionized water

$$\mathrm{I}, \% = 100 - \frac{\mathrm{Ls} \cdot 100}{\mathrm{Lc}},$$

where Ls — average luminescence in the sample supernatant and Lc — average luminescence in water.

7.3.3.1. Bioluminescent direct contact flash test

The bioluminescent direct contact flash test is a modification of the direct contact luminescent bacterial test described below. In this method, the kinetic measurement of luminescence is started at the same time the *V. fischeri* suspension is added to the sample (5 g of the solid sample is suspended with 45 ml of 2% NaCl by mixing vigorously for 5 min). Luminescence signal is measured 20 times per second throughout the 30-s exposure period. Peak height at 0–5 s (I_0) and signal at 30 s (I_t) were measured. The result is expressed as the ratio of I_t and I_0, and converted to inhibition percentage (I%).

The results are compared with the inhibition caused by 2% NaCl, which is used as a clean control sample. The procedure is carried out with luminometer at 20°C and with the BioTox kit containing freeze-dried *V. fischeri*.[9]

7.3.4. *Toxi-Chromo-Pad Test*

The Toxi-Chromo-Pad test is a biotest based on the inhibition of β-galactosidase enzyme synthesis in *E. coli* when the bacterial cells are directly exposed to contaminated samples.[97]

The Toxi-Chromo-Pad test with soil samples is carried out according to the manufacturer's instructions.[98] Add freeze-dried rehydrated *E. coli* to the reaction mixture. After 20 min preincubation, mix 0.5 ml of this solution with 0.5 g of homogenized fresh soil. Incubate the mixture at 37°C for 2 h. Homogenize the mixture again and pipette a small drop of incubated soil bacterial mixture into a chromogenic pad (a glass microfiber filter soaked with blue chromogenic substrate for β-galactosidase). Incubate the pads at 37°C overnight for color development. The intensity of enzyme synthesis is indicated by the color blue; possible toxicity reduces enzyme synthesis and color development.

7.3.5. *MetPlate*

The MetPlate bioassay provides a sensitive indication of the presence of heavy metal toxicity. This test is based on the inhibition of the biosynthesis of β-galactosidase in *E. coli*. Bacterial suspensions are prepared by rehydration of lyophilized cells in 3.0 ml ultra-pure water.

Weight 1 g samples of sieved soils or wastes in test tubes and mix by vortexing with a volume of 0.1 ml bacterial suspension and 0.9 ml ultra-pure water. Incubate the mixture at 35°C for 60 min. Add subsequently 0.5 ml of chromogenic substrate (red chlorophenol β-*d*-galactopyranoside) to each tube. Incubate the tubes at 35°C until development of a purple color. The intensity of the resulting purple color gives an indication of β-galactosidase activity and is inversely proportional to the sample toxicity. Filer the mixture from each tube . Measure absorbance of 0.4μm of the solution at 575 nm using a microplate reader. Blanks are run with ultra-pure water instead of soil or waste suspension. All tests are performed in triplicate.[3]

Use 1 ml of water instead of 1 g sample for the control.

$$\mathrm{I}, \% = 100 - \frac{\mathrm{As} \cdot 100}{\mathrm{Ac}},$$

where As — average absorbance in the sample and Ac — average absorbance in control.

7.3.6. Bacillus Cereus *Dehydrogenase Activity Test*

Inoculum of *B. cereus* is prepared from the actively growing biomass. Optical density (by 600 nm) is adjusted to 0.5 ($1.5^{*}10^{8}$ cells/ml) and 2-ml cell suspension are added to a solid-water suspension in a screw-cap tube containing 2 g soil (dry) and 4 ml distilled water. Tubes are then shaken overhead at 70 rpm for 2 h by 25°C. 4 ml rezazurin solution (30 mg rezazurin dissolved in 1l potassium-phosphate buffer) is added. After one hour incubation (by overhead rotation described above), sampled are centrifuged by 10,000 rpm for 5 min. The clear supernatant is measured photometrically at 600 nm for unreduced rezazurin. Chemical activity of the soil or reaction between dye and contaminants is determined by subjecting inoculum to the same water volume (blanks).[7]

$$\mathrm{I},\% = 100 - \frac{(\mathrm{Ds_b} - \mathrm{Ds}) \cdot 100}{(\mathrm{Dc_b} - \mathrm{Dc})},$$

where Ds — average optical density in the sample, $\mathrm{Ds_b}$ — average optical density in the sample blanks, Dc — average optical density in control, and $\mathrm{Dc_b}$ — average optical density in control blanks.

7.3.7. *Contact Test with Three Soil Animal Species*

Three different soil animal species (*Folsomia candida*, *Enchytraeus albidus*, and *Enchytraeus* sp.) are used for testing. Soil and waste samples are moistened with deionized water and mixed carefully until a moist and crumby structure is achieved. 20 g (dry weight) of each sample is weighed into 10 glass beakers (50 ml) and into five glass jars (250 ml). Ten *F. candida* specimens are transferred into five beakers, 10 *Enchytraeus* sp. specimens into another five beakers, and 10 *E. albidus* specimens into five glass jars. Small amounts of baker's yeast for the collembolans or grinded oat flakes for the enchytraeids are given as food. Finally, the beakers are closed with a plastic film (paraffin) that is pierced for gas exchange, and the jars are closed with a glass lid. The vessels are incubated in a climate chamber (20°C, constant light 400–800 lx) for 32 days. Evaporated water is compensated and food is added once a week. After 14 days, *E. albidus* adults have to be removed from the soil or waste and counted. After 32 days, the soils in the jars are wetted with ethanol (99.5%) and *ca.* 10 drops of Bengal red (1% ethanol solution) are added into each jar. The juveniles are hand sorted and counted after dying. Collembolans are extracted from the soil samples with a high-gradient extractor. Enchytraeids are extracted with the wet funnel method where temperature was raised with glow bulbs for 4 h. Worms are stained in ethanol and then dyed Bengal red before counting. For all species, adults and juveniles have to be counted separately. Artificial soil is used as a control. All tests are performed in three replicates.

For each waste or soil, six inhibitions are calculated: for each of three organisms, adults and juveniles separately. Depending on investigation goals, Inhibitions can be presented separately or average inhibition of six parameters can be counted.

In each of six cases, inhibition is calculated as following:

$$\mathrm{I},\% = 100 - \frac{\mathrm{Ns} \cdot 100}{\mathrm{Nc}}$$

where Ns — average number of worms in the sample and Nc — average number of worms in control.

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Chapter 13

GROUNDWATER CONTAMINATION AT LANDFILL SITE

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Abstract

Improper design of landfill sites may cause serious problems to groundwater. Leachate, a very concentrated pollutant generated from the decomposition of waste and by precipitation, may penetrate through the waste layers and go straight to the aquifer. This chapter discusses some background on groundwater resource, its properties, and monitoring at landfill sites. Pulau Burung landfill in Penang, Malaysia was taken as the case study site where boreholes and monitoring wells were sampled and the results discussed.

Keywords: Groundwater, landfill, borehole, monitoring well, leachate.

1. Introduction

In most countries, the landfill is still the preferred method for solid waste disposal due to its low cost and maintenance. As the volume of waste produced grew, more landfill sites were built without increasing waste disposal costs. Because of their large numbers and nature of operations, landfill sites are now considered a threat to the environment, especially those that are not properly maintained and controlled. One of the threats posed by landfill sites is the production of leachate, which is generated from the decomposition of waste and by moisture that penetrates the landfill layers. Leachate is a highly polluted wastewater that contains a wide range of contaminants including organic and inorganic matters, humic and fulvic substances, and even heavy metals. The discharge of untreated leachate into a watercourse could lead to serious pollution of the water supply. This is also a prolonged problem since leachates continue to be produced for many years even after a landfill site has been closed.[24]

For some time now, there has been an increased concern regarding the contamination of groundwater by leachate infiltration. The uncontrolled infiltration of leachate from the unsaturated zone into the saturated zone (groundwater) is considered the worst environmental impact of landfills. Infiltration of leachate is a common phenomenon in an open and unlined dump site without a leachate collection system. In 1977, a study on 50 landfill sites discovered the presence of organic chemical contamination at 40 sites and infiltration by at least one hazardous chemical at 43 sites. Groundwater contamination in a landfill site depends on factors such as the composition of leachate, operation of landfill, and interaction of contaminant transport in the soil.[23]

To prevent infiltration of leachate into the groundwater, methods such as leachate collection systems, liners, and also capping are integrated into the design of a landfill site. Landfills using these types of waste disposal systems are referred to as sanitary landfills. These engineered waste disposal systems employ spreading, compacting waste, lining system, and daily soil covering. This chapter discusses how a groundwater-monitoring program can be used to effectively control the contamination of groundwater from leachate at landfill sites by focusing on the practice used by the Pulau Burung landfill site in Penang, Malaysia. For assessment and selection of groundwater monitoring at a landfill site, an understanding of the contaminant source (landfill leachate), contaminant transport (hydro geological), and receptor (groundwater) is necessary. Table 1 summarizes the aspects of a landfill site that must be considered before implementation of a groundwater monitoring network.

Table 1. Landfill Aspects for the Selection of a Groundwater Monitoring System.[1]

Landfill site	(1) Site characterization (area, waste cell structure, and depth) (2) Landfill operational system (compaction, liner, cover, and waste disposal rate) (3) Waste type and properties (composition, permeability, and density) (4) Leachate characteristics (5) Liner properties (possibility of leakage mechanism — diffuse and tear) (6) Surface run-off
Hydrogeology	(1) Groundwater system (unsaturated zone, saturated zone, and vadose zone) (2) Geology characteristic (soil type, porosity, and permeability) (3) Hydraulic (flow rate, direction, and quantity) (4) Contaminant transport (pathways and attenuation)
Rainfall	(1) Rainfall statistics (2) Catchments area (up gradient and down gradient)

2. Groundwater Resource

2.1. *Groundwater Aspects*

Generally, groundwater is the subsurface water that penetrates through pores and cracks in the soil. Several terms regarding groundwater are given in Table 2. Under normal conditions, the land surface or the unsaturated zone is sufficient for protecting the quality of groundwater below. However, due to the disruptive existence of landfill sites, the natural processes or attenuation mechanisms that convert contaminants on the surface into harmless matter are no longer effective. Because of this contaminants must be treated before they can penetrate the ground and reach the water table.

2.2. *Groundwater Flow Rate*

In an unsaturated zone, the groundwater flow is mostly gravity driven. The horizontal flow in a saturated zone depends primarily on the porosity of the soil. There are three distinct flow mechanisms:

1. *Intergranular flow*: A flow through evenly distributed and interconnected pores.
2. *Fissure flow*: A flow through fissures that occurs in the absence of pore openings (low-porosity and high-permeability conditions). This flow is less predictable, but more rapid than intergranular flow.
3. *Flow in conduit*: A flow that is channeled through a conduit. This flow is as high as a surface flow; therefore, attenuation would be insignificant but dilution might occur.

Table 2. Groundwater Terms.[2]

Terms	Explanation
Unsaturated zone	A zone that exists between the water table and the surface. This zone is only partially filled with water since it consists of an opening, pores, and crack in the soils. Water passes through this surface into the water table layer below and its movement is gravity driven.
Saturated zone	A zone in which all the pores, openings, and cracks in the soils are filled with water. Typically, it is located in the upper layer of bedrock in which water cannot penetrate.
Water table	This is the top layer of the saturated zone. It changes (rise or fall) depending on the season, rainfall, and snowmelt. The pattern of changes is usually the same throughout the year except during heavy rainfall or drought.
Aquifers	This is a water-bearing soil or rock formation that is capable of both storing and transmitting a certain amount of water.
Permeability	It is the measurement of the speed of water flowing through an opening or the pores in the soil.
Porosity	This is the measured capacity of soil to contain or hold water. Usually saturated sand contains 20% water, 25% for gravel, and 48% for clay.

The flow rate of groundwater can be calculated using the simple groundwater equation:

$$Q = K \times i/n \tag{1}$$

where $Q =$ flow rate (m/s), $K =$ hydraulic conductivity (m/s), $i =$ hydraulic gradient (m/m), and $n =$ effective porosity (dimensionless).

3. Landfill

3.1. *Operation at the Pulau Burung Landfill Site*

The Pulau Burung landfill site is situated in the Byram Forest Reserve (5° 24′ N, 100° 23′ E) and covers an area about 62.4 hectares wide. In terms of location, this site has proven to be a very valuable research site since it is located within a mangrove area and near the sea. Perimeter drains surround the landfill area. During its early operation in the 1980s, the landfill site was used merely as an open-dumping site for municipal solid waste (MSW). The waste disposal operation for the landfill site was managed by the Seberang Perai Municipal Council. In its first 10 years of operation from the early 1980s until 1990, waste was disposed without proper management and improper leachate control. During the time (1980–1990), waste disposed at the site came from Seberang Perai Selatan area and from commercial and industries areas in Seberang Perai Utara.

In 1991, the landfill site was upgraded to a Level II sanitary landfill site by implementing a semi-aerobic system. This system, also known as the Fukuoka Landfill

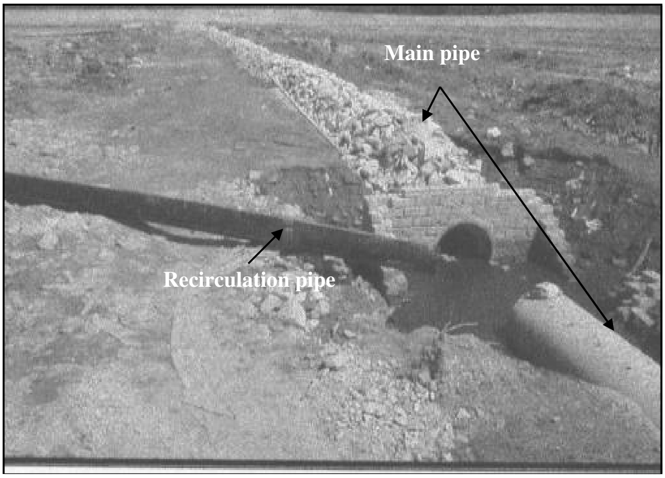


Figure 1. Constructed Main and Recirculation Pipe at Pulau Burung Landfill Site.[6]

Method, was developed by Fukuoka University in 1975. This method of land filling allows the movement of air into the waste layer through a network of horizontal pipes at the bottom of landfill site and a vertical gas vent pipe.[3] Movements of air are due to convection caused by a temperature differential between the waste layer and the atmosphere. After management of the landfill site was given to Idaman Bersih Sdn. Bhd. on August 1, 2001, it was further upgraded to a Level III sanitary landfill by implementing the recirculation of leachate into the waste layer.[4]

Presently, an average of 1,800 tons of solid wastes is being disposed of daily at this site.[5,25] It now disposes solid waste (both municipal and nonhazardous industrial wastes) from Seberang Perai Selatan, Seberang Perai Utara and Penang Island. The site is also equipped with other control systems including a leachate collection system, a gas collection system, and aeration for a leachate collection pond. Figures 1 and 2 show the construction of a horizontal leachate collection pipe at the landfill site. Two categories of waste received by the site are flammable waste and organic industrial waste. The landfill site is also of interest since it is located near the sea. A river is also nearby, and a constructed drain (perimeter drain) surrounds the landfill area. Figure 3 shows the leachate collection pond at the landfill site and Fig. 4 shows the schematic diagram of the Pulau Burung landfill site area.

3.2. *Landfill Type and Design*

Many types of landfills are used for the disposal of different type of wastes, whether from industries or domestic waste streams. The Pulau Burung landfill site is a co-disposal landfill site, as it accepts various wastes from residential areas and

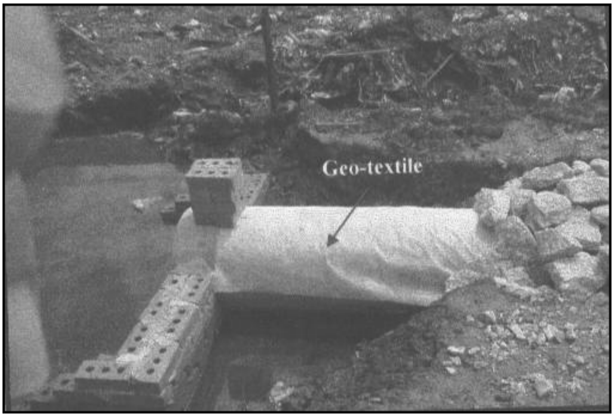


Figure 2. Geotextile are Installed to Prevent Blockage of Pipeline.[6]

Figure 3. Leachate Collection Pond (End Line of Main Leachate Pipeline).[7]

nonhazardous wastes from industries. These types of landfills play important roles in determining the groundwater-monitoring system since it affects the characterization of leachate produced, the design of the landfill layer and thus the degree of risk for contamination to the subsoil. Table 3 summarizes certain types of landfills and their descriptions.

3.3. *Landfill Design and Liner System*

Liner systems are used within containment landfills as barriers to prevent an outflow of leachate into the surrounding soil. Liners implemented by landfill operators can be either geosynthetic or natural. Most landfill operators prefer natural liners to geosynthetic ones because of their lower cost. A natural liner is compacted at the

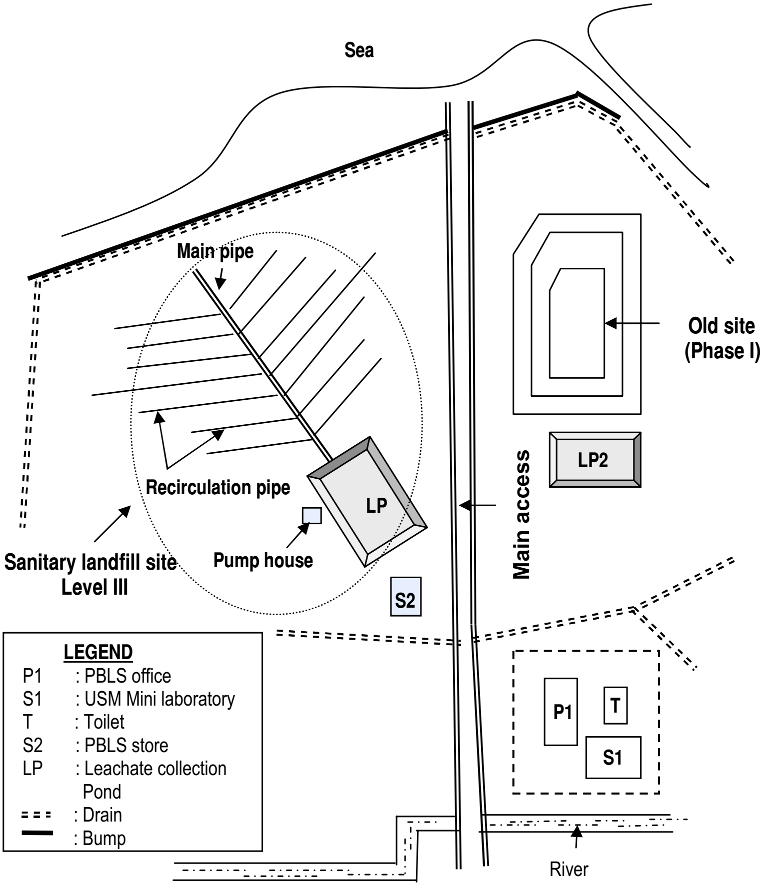


Figure 4. Schematic Diagram of Pulau Burung Landfill Site.[6]

base of the landfill, and natural soil, usually rich in clay or bentonite, is used as a liner. Bentonite is used since it is a material of low hydraulic conductivity and, thus, low permeability.[8] Bentonite is also extremely hydrophilic and attracts water between its layers, leaving little void/space. In certain cases, geosynthetic clay liners, which consist of a combination of bentonite with geotextile or geomembrane, are used, and combinations with zeolite are believed to prevent the transport of phenolic compounds.[30] Most sodium bentonites used as geosynthetic clay liner have a permeability around, and a combination with zeolite are believed to prevent phenolic compounds transport[30] 1×10^{-9} to 5×10^{-9} cm/s.[8]

Even though a liner in landfill design may be installed, infiltration of leachate into the soil is still possible with the likelihood of leachate management systems failing.[27,29,31] Infiltration of leachate may occur due to leakage in the liner design and the limitation of the liner's lifespan compared to the lifespan of leachate production,

Table 3. Type of Landfill and Description.[1]

Type of Landfill	Description
Dilute & disperse	This site does not provide any barrier between the waste and the soil surface. The leachate is allowed to seep into the soil layer and depends on natural attenuation to reduce contaminant concentration.
Bioreactor	It is designed for the purpose of maximizing the degradation rate of waste disposed at the site. This is done through the recirculation of leachate onto the waste layer.
Containment	Designed to minimize the inlet and outlet of moisture from the landfill site. Usually equipped with a leachate collection and management system.
Noninert	It can be a mono-disposal site, co-disposal site, or liquid waste site. For mono-disposal, the leachate produced will have similar characteristics as the waste type. A co-disposal site accepts both solid and liquid wastes; thus, it must be monitored and engineered to prevent contaminants due to its leachate.
Inert	This landfill site only disposes of inert wastes. Therefore, it would not produce a significant volume of leachate but there is still a risk due to surface run-off that could wash the waste away. Incoming wastes have to be controlled very well.

which extends for years after the landfill site is closed. Leachate may exit the landfill layer toward the subsoil either by diffuse or discrete leakage. Discrete leakage would be hard to monitor since it could be caused by a tear on a membrane or geosynthetic liner and would be difficult to detect. Meanwhile, diffuse leakage usually occurs if natural or mineral liners are used and it spreads on a wider scale. Therefore, the monitoring points can be separated widely within the diffuse area for detection of contamination of groundwater. Infiltration of leachate through diffuse leakage also would be less significant since attenuation of contaminants could occur while it diffuses through the liner layer. In the case of the Pulau Burung landfill site, a natural liner consisting of natural marine clay is laid at the base of the landfill site.[9,22] The main component of the natural liner consists of a low permeability clay that prevents the outflow of leachate from the landfill site and also the inflow of groundwater into the waste layer, retarding the transport of leachate solutes.[26]

3.4. *Soil Properties*

Soil properties will determine the permeability, porosity, and also the attenuation mechanism that would affect the movement of contaminants through the soil. In short, it plays a very important role in determining the movements of leachate as it seeps out from the landfill layer and the fate of contaminants as it flows down the subsoil into the groundwater below. A subsoil may consist of different types of soils

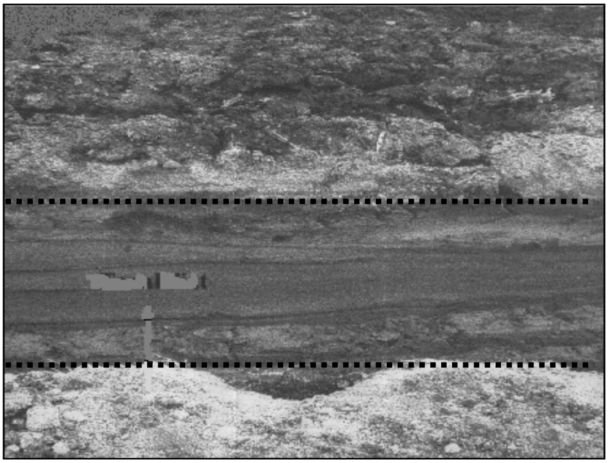


Figure 5. Soil Layer Under Leachate Collection Pond at Pulau Burung Landfill Site.[6]

and thus have different properties at each layer. An example of subsoil properties at the Pulau Burung landfill site is shown in Figs. 5 and 6. It is, however, difficult to quantify the porosity and permeability of the soil mixture. Table 4 shows the porosity and permeability of different soil types and their specific yield (ratio of the water volume to the rock volume).

4. Leachate

Parameters that need to be identified for landfill leachate are the leachate generation rate and the leachate concentration. Knowledge about the generation rate of leachate is important especially for designing the leachate pipes, leachate collection systems, the retention pond, leachate treatment facilities, etc. Landfill leachate is produced from a number of sources. The main sources of landfill leachate include: water produced from the decomposition of waste in the landfill layers, water squeezed out due to the waste weight, and water that passes through the waste, especially during rainfall. Landfill leachate will flow to the bottom and accumulate beneath the landfill site. Landfill leachate contains a wide range of contaminants, both organic and inorganic; some of these contaminants, such as biochemical oxygen demand (BOD), chemical oxygen demand (COD), and ammoniacal nitrogen, are highly concentrated. Table 5 provides characteristics of landfill leachate at the Pulau Burung landfill site in 2003.[11]

4.1. *Leachate Generation*

The production of landfill leachate is affected by factors such as the types of waste disposed at the site, landfill age, and amount of rainfall. These factors not only determine

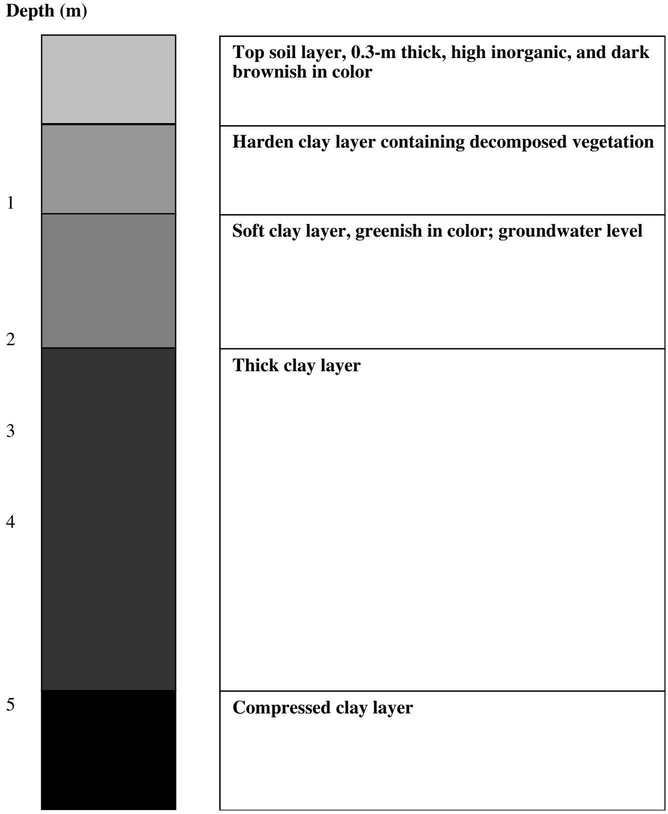


Figure 6. Diagram of Soil Layer at Pulau Burung Landfill Site.[9,10]

Table 4. Typical Porosity, Permeability, and Specific Yield of Different Type of Soils.[2]

Soil Material	Specific Yield (%)	Porosity (%)	Permeability (m/day)
Clay	1–5	45–55	1×10^{-8} to 1×10^{-4}
Sand	10–30	25–45	1×10^{-1} to 1×10^{3}
Gravel	15–30	25–35	1×10^{3} to 1×10^{5}
Sand and gravel	10–20	20–30	1×10^{1} to 1×10^{3}
Silt	5–10	35–50	1×10^{-4} to 1×10^{1}

the quantity of leachate produced but also affect its composition and quality. As the layer of waste increases on a landfill site, more moisture within the waste will be released due to the rainfall that seeps into the layers, in addition of the weight of deposited waste layer. Therefore, in humid areas such as Malaysia, the amount of leachate produced is quite high due to the high precipitation that exceeds evaporation. A study was conducted at two sanitary landfill sites in Malaysia, Ampang

Table 5. Leachate Parameter at Pulau Burung Landfill Site in 2003.[11]

Parameter	Jan	Feb	March	April	May	June	July	Aug	Sept	Oct	Nov	Average
pH	7.89	8.18	8.42	8.96	8.78	8.44	8.2	8.05	8.84	7.91	8.39	8.37
BOD_5 (mg/L)	75	98	85	66	140	181	207	548	605	1120	345	315
COD (mg/L)	1640	1700	1702	1656	1717	2338	2070	2118	2393	3450	2197	2089
Color (PtCo)	2430	2530	2940	3070	3264	3360	3680	3560	3400	6540	3630	3491
Turbidity (NTU)	174	104	67	80	100	110	114	160	150	450	170	153
SS (mg/L)	219	220	164	196	210	251	227	303	340	936	380	313
Iron (mg/L)	5.5	5.1	5.6	3.5	4	5.9	5	3.4	4.6	8.6	6.2	5.2
Ammoniacal nitrogen (mg/L)	1909	429	1017	830	626	508	820	1201	1724	1325	1104	1044.8
Oil and grease (mg/L)	7	15	10	41	3	8	9	15	13	17	18	14.2
Zinc (mg/L)	0.5	0.4	0.4	0.5	0.1	1.5	0.6	0.6	0.8	1.8	0.8	0.7
Nickel (mg/L)	0.4	0.3	0.2	0.1	0.1	0.2	0.1	0.1	0.1	0.1	0.2	0.2
Tin (mg/L)	0.1	0.1	0.1	0.1	0.1	0.2	0.1	0.2	0.1	0.1	0.2	0.1
Cadmium (mg/L)	0.04	0.04	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02	<0.02

Jajar landfill site, and Pulau Burung landfill site in Penang, to quantify the actual amount of leachate generated in the field from different ages of landfill. It was found that the maximum leachate generation rate for a four-year-old landfill was about $8\,\text{m}^3{\cdot}\text{lift}^{-1}{\cdot}\text{ha}^{-1} \cdot \text{day}^{-1}$ and named SALL4. For a 10-year-old landfill, the leachate generation rate was about $4.2\,\text{m}^3 \cdot \text{lift}^{-1} \cdot \text{ha}^{-1} \cdot \text{day}^{-1}$ and named SALL10.[12]

Some fractions of leachate are lost due to evaporation, evapotranspiration, and absorption by the waste. The evaporation of leachate depends on the factors such as the specific gravity of the leachate, ambient temperature, atmospheric pressure, wind velocity, and the difference of vapor pressure between the atmosphere and the waste surface. Evaporation also depends on the presence or the availability of the leachate to be evaporated. Evaporation rate decreases about 1% for every increase of 1% in specific gravity.[13] Evapotranspiration is the combination of evaporation and transpiration. Evaporation occurs on the soil surface while transpiration refers to the loss of water due to uptake by plants. Therefore, losses of leachate through transpiration are more significant than evaporation since transpiration processes reach deep down to the depth of plant roots.[8]

Leachate is absorbed by both the soil and the waste. The amount of leachate absorbed by soil depends on the moisture content and especially the soil's specific yield, which is shown in Table 4. Absorption of leachate by both soil and waste only occurs until the field capacity of the waste and soils has been reached. After this point, all the liquid in the waste layer will flow down as leachate. The field capacity is the maximum moisture content that can be retained by waste and soil against gravitational force. Thus, the absorption of waste depends mostly on the composition of waste and its moisture contents. The field capacity of refuse in the landfill site can be estimated if the relative percentage of each type of waste component is known. For MSW, a field capacity of 33 cm/m is considered reasonable with an initial waste moisture content of 12 cm/m.[13] Figure 7 shows the infiltration of rainfall as it moves into the waste layer.

4.2. *Leachate Composition*

4.2.1. *Waste Type*

The varying moisture content of different types of waste affects the quantity and quality of leachate. Table 6 summarizes the types of waste dumped at the Pulau Burung landfill site. Moisture inside the waste will seep downward as the waste is being compressed or due to increased weight as additional layers of waste are being dumped and compacted. As the layer of waste increases, the water that flows down will also bring contaminants along with it, as it passes through the layers of waste. Knowledge of the variation of wastes dumped at a landfill site is important

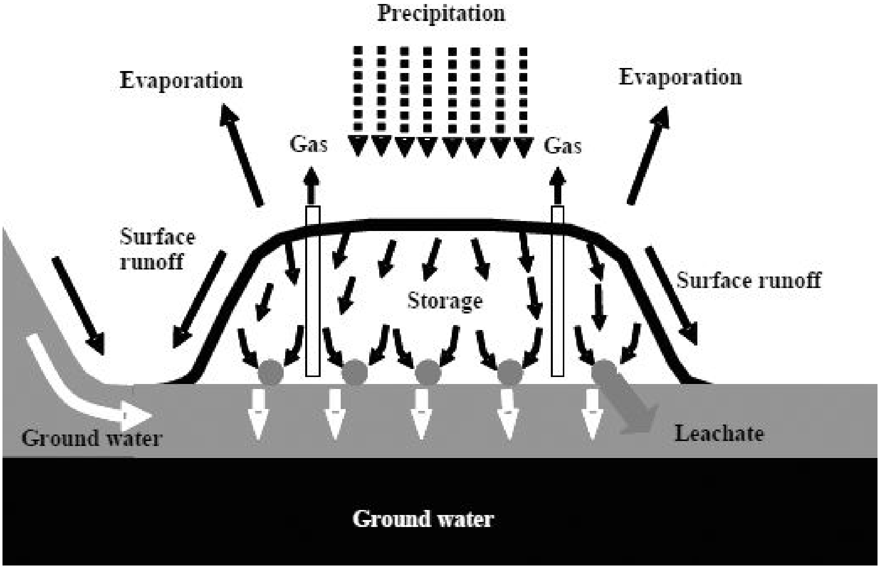


Figure 7. Hydrology at Landfill Site.[14]

Table 6. Waste Disposed at Pulau Burung Landfill Site in 2003.[7]

	Solid Wastes										
	Domestic		Garden		Construction		Industries		Other		
Month	Trips	%	Trips	%	Trips	%	Trips	%	Trips	%	Total
January	1482	19.1	66	0.9	29	0.4	5816	75.0	363	4.7	7756
February	1304	22.1	56	0.9	6	0.1	4236	71.8	301	5.1	5903
March	1405	19.3	85	1.2	35	0.5	5422	74.4	344	4.7	7291
April	1402	19.5	59	0.8	21	0.3	5335	74.4	358	5.0	7175
May	1575	21.7	119	1.6	19	0.3	5179	71.4	362	5.0	7254
June	1402	20.5	86	1.3	21	0.3	5001	73.0	343	5.0	6853
July	1462	20.0	102	1.4	13	0.2	5331	73.1	384	5.3	7292
August	1419	20.2	118	1.7	23	0.3	5097	72.5	370	5.3	7027
September	1398	19.5	79	1.1	22	0.3	5314	74.2	347	4.8	7160
October	1664	23.1	89	1.2	17	0.2	5022	69.8	399	5.5	7191
November	1651	24.8	74	1.1	16	0.2	4638	69.6	288	4.3	6667
December	—	—	—	—	—	—	—	—	—	—	—
Total	16164	20.8	933	1.2	222	0.3	56391	72.7	3859	5.0	77569

for determining the type of contamination that may occur at the site. It also provides very useful information regarding the volume of leachate generated.

4.2.2. *Landfill Age*

As a landfill ages, the waste in the landfill layer will degrade and result in different by-products. Phases of a degradation process inside the landfill layer are aerobic, anaerobic, methanogenic, fermentation, etc. These processes are influenced by microorganism activity between the waste layers as well as the availability of oxygen. In the early stage, decomposition of waste in landfill layer occurs under aerobic conditions since oxygen is still present within the compacted waste layers. However, this condition only lasts for a few days or weeks. This stage produces heat as high as 80–90°C and a complex solution within a neutral pH value. Not much is contributed to the leachate composition at this point, but the high temperature generation is very important in enhancing the waste decomposition. Once the oxygen has been fully utilized within the waste layer, decomposition occurs anaerobically. Leachate production during this stage will contain high concentrations of ammonia, metals and soluble degradable organics, and is very acidic. After several months or years, leachate will be produced under methanogenic conditions. Leachate will be in a slightly alkaline or neutral condition with much lower contaminant concentrations but still high in ammonia. Once biodegradation is nearly completed, aerobic conditions prevail. Table 7 shows the typical composition of leachate at different ages of landfill.

4.2.3. *Rainfall Statistics*

Rainfall is the means by which outside water infiltrates the landfill system and causes the amount of landfill leachate to increase. Even though the infiltration of rainfall into the landfill increases the quantity of leachate, it also changes the concentration of leachate through dilution processes. Table 8 shows the rainfall statistics for the Butterworth station located about 30 km from Pulau Burung landfill site. An example of the effect caused by rainfall on the leachate quality (color and COD) at Pulau Burung landfill site is shown in Fig. 8.

4.3. ***Infiltration of Leachate***

Leachate will flow downward to the bottom of the landfill and if no liner is present, will continue to seep downward until it reaches the water table. The availability of a liner in a landfill will minimize or attenuate the flow, and leachate collection pipes will direct the flow to a designated area (leachate collection pond) prior to treatment before discharge. However, there have been few studies on the possibility of failure of an a liner system to prevent the infiltration of landfill leachate into the groundwater.

Table 7. Typical Data on the Composition of Leachate From New and Matured Landfill.[15]

	Value, mg/L		
	New Landfill (Less than 2 Years)		Mature Landfill (Greater than 10 Years)
Constituents	Typical	Range	
BOD_5	10,000	2,000–30,000	100–200
TOC	6,000	1,500–20,000	80–160
COD	18,000	3,000–60,000	100–500
Total suspended solids	500	200–2,000	100–400
Organic nitrogen	200	10–800	80–120
Ammonia nitrogen	200	10–800	20–40
Nitrate	25	5–40	5–10
Total phosphorus	30	5–100	5–10
Ortho phosphorus	20	4–80	4–8
Alkalinity as $CaCO_3$	3,000	1,000–10,000	200–1,000
pH	6	4.5–7.5	6.6–7.5
Total hardness as $CaCO_3$	3,500	300–10,000	200–500
Calcium	1,000	200–3,000	100–400
Magnesium	250	50–1,500	50–200
Potassium	300	200–1,000	50–400
Sodium	500	200–2,500	100–200
Chloride	500	200–3,000	100–400
Sulfate	300	50–1,000	20–50
Total iron	60	50–1,200	20–200

Even in an engineered landfill site, there are still possibilities of leakage due to design faults or limitations, or the exceeding of a liner's lifetime. Therefore, an acceptable leakage limit typically considers the physical and attenuation properties of the lining system.

4.4. *Attenuation of Leachate*

Contaminant transport at a landfill site involves numerous movements and changes, such as the passing of contaminants through the liner to the groundwater flow and the discharge of groundwater to surface water. These changes are due to the attenuation properties[28] of either the lining system or the soil type at the site. Attenuation causes a decrease in contaminant concentration as it undergoes physical, chemical, and biological processes such as sorption and desorption, precipitation and dissolution, advection/ dispersion, dilution, and degradation. Figure 9 shows an example of the attenuation of leachate as it infiltrates into the groundwater system, and Table 9 summarizes the parameters affected by these attenuation processes.

Table 8. Rainfall Statistic at Butterworth Station.[7]

	August 2003	September 2003	October 2003	November 2003
Days	Rain (mm/day)	Rain (mm/day)	Rain (mm/day)	Rain (mm/day)
1	31.1	9.1	89.8	1.2
2	44.3	13.7	82.4	2
3	20.9		113.6	8.4
4	T		123.7	4.1
5			82.5	T
6			5.7	1.1
7		82.8	3.9	1.2
8		70.5	9.9	27.6
9	0.2	5.2	T	5.2
10	T	T	0.2	22.3
11		4.9	T	18.1
12	4.9	8.6	0.2	4.1
13	15.3	23.5	10.5	
14	46	22.4	16.1	
15	38	0.1		32.4
16	13.6		8.6	1.5
17	6.5		2.3	
18		34.2	7.3	0.6
19		21.9	22.8	14.6
20		1.3	60.6	5.8
21		0.2	7.8	
22			3.7	2.6
23	34.8	6.9		6.5
24		0.8	9.9	T
25		6	0.1	22.1
26		0.2		20.4
27	38.2	12.8		
28	0.3	0.4		34.1
29	20.9		2.2	
30	17.2		6.9	
31	1.7		3.1	
Total	333.9	325.5	675.4	235.9

4.4.1. *Adsorption*

One of the major attenuation processes[29] occur within the subsurface soil and involves the interaction of contaminant toward the soil surface due to either physical attraction (physical adsorption) or chemical bond (chemisorption). Both interactions are almost the same as they are the processes by which dissolved contaminants are prevented from entering the aqueous phase, and they will be referred to here simply as the adsorption process. The process occurs at the surface of a certain solid matter

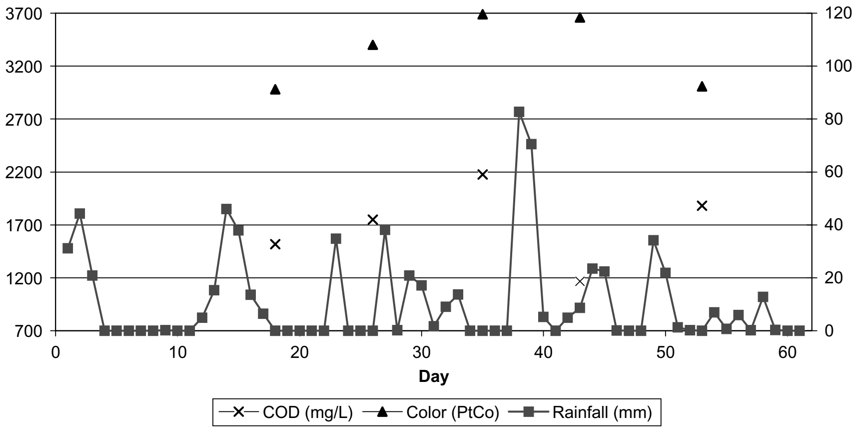


Figure 8. Relationship of Rainfall (August–September 2003) for COD and Color in Leachate.[7]

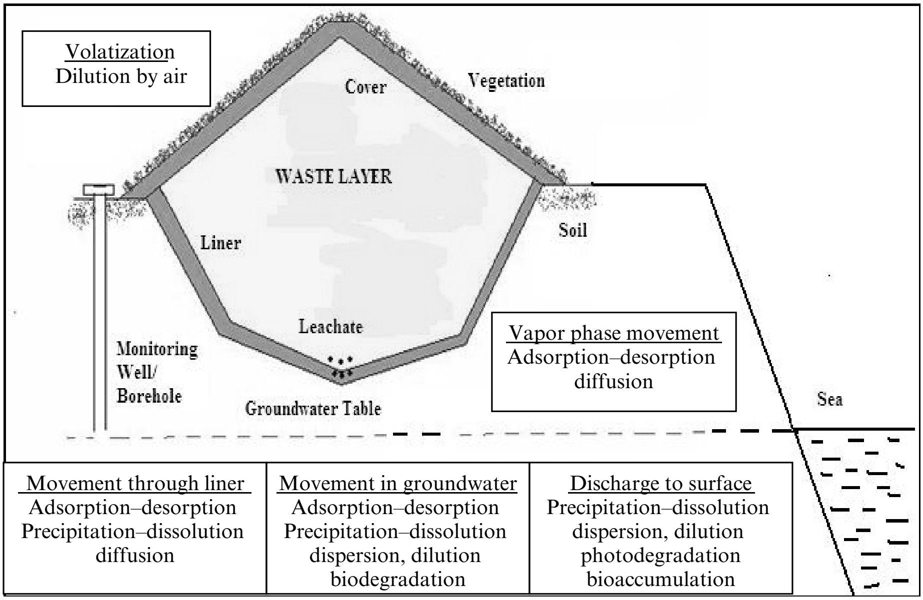


Figure 9. The Attenuation of Leachate.[16]

and the aqueous layer with which it comes in contact. This process is also sometimes referred to as an adsorption-exchange. Adsorption is known to cause a decrease in leachate TDS value, while the exchange only changes the type of ion in the leachate but does not reduce the value of TDS.[1,17] The adsorption process is induced by a chemical process where the charged ions of contaminants are attracted to the surface or into the solid interior. The processes are either complete or partially reversible.

Table 9. Attenuation Processes and Affected Parameters.[13]

Attenuation Processes	Parameters Affected
Precipitation/dissolution	Al, As, Ba, Br, Cd, Ca, Cr, Cu, Fe, Pb, Mg, Mn, Hg, Ni, K, Se, SiO_2, $SO_{4\ 2-}$, Zn
Sorption/desorption	As, Ba, Br, Cd, Cr, Cu, Fe, Pb, Hg, Ni, Zn
Complexation	Ba, Ca, Cr, Cu
Ion exchange	NH_3, Ba, Ca, Cr, Cu, F^-, Fe, Pb, Mg, Mn, K, Se, Na, $SO_{4\ 2-}$, Zn
Filtration	COD (minor)
Dilution	Cl, VOC
Degradation	NH_3, COD, Fe, NO_{3-}, VOC

Common adsorbents in the soil are sand, clay, organic matter, humic substance, and also hydrous metal oxides.[18]

4.4.2. *Complexation*

Complexation occurs between complexing agents and metal ions to form complex matter (soluble or insoluble). The formation of complexes will affect the dispersion of contaminants in the groundwater body. Typical complexing agents are sulfide, sulfate, chloride, and bicarbonate species. The complexation process depends on pH, DO, metal, and anion concentration.[18]

4.4.3. *Precipitation/Dissolution*

In this process, contaminants either enter the aqueous phase or form a solid phase. It is the same phase as a complexation process where complexing agents react with ions of metals to form soluble or insoluble complexes. These processes are affected by the factors such as the pH value, temperature, pressure, type, and concentration of complexing agents and also the oxidation agents.[18] This process affects the extent of trace metal contamination in the groundwater and also the mobility of the metals in the leachate contaminated groundwater plume.

4.4.4. *Advection/Dispersion*

Advection/dispersion occurs because of the motion or flow of groundwater. This motion enhances the spreading of contaminants into the groundwater and influences the extent of the contamination zone. Dispersion spreads the contaminants to a larger area in the groundwater whereas the advective transport of contaminant only affects a certain point in a groundwater aquifer. The dispersion of contaminants in the groundwater is similar to the dispersion of gases in the atmosphere. It is also affected by the groundwater velocity. Many researchers also believe that the dispersitivity also depends on the scale of area being studied.

4.4.5. *Dilution*

Dilution of the concentration of contaminant is accomplished through contact with existing water bodies or the addition of water. This process does not chemically alter the leachate composition. Dilution occurs as soon as the leachate-concentrated liquid comes into contact with other water sources. In this case, the water source is formed either from rainfall that seeps into the ground or from the existence of groundwater in the soil. Dilution reduces the concentration of contaminant that is present in the highly-polluted leachate. Hardness, chloride, nitrate, and sulfate that are not attenuated by soil will be reduced by the dilution process.[13] This process is influenced by a difference of density between leachate and groundwater, groundwater velocity, diffusion/dispersion coefficient, soil strata, leachate entry velocity, and the landfill area.[13]

4.4.6. *Filtration*

Filtration involves the physical entrapment of suspended solids and settleable solids in leachate. The efficiency of the process depends on leachate hydraulic gravity and soil pore size. However, the change in leachate properties due to this process is quite insignificant.

4.4.7. *Degradation (Biodegradation, Hydrolysis, and Oxidation-redox)*

Biodegradation is a process in which organic compounds are oxidized by microorganisms. Active microbial populations are present in the subsurface layer. Biodegradation is limited by numerous factors including the concentration of the contaminants, pH, presence of toxicants, adsorption, temperature, as well as the lack of oxygen and other nutrients such as phosphorus and nitrogen. Hydrolysis involves the breakdown of mineral by H^+ and OH^- that are present in the groundwater. This process is influenced by the pH, type and concentration of cations, oxidation-redox state, and temperature. The hydrolysis process is enhanced at a high temperature, low pH, low oxidation-redox state, and with low organic contents. Most highly charged trace metals are strongly hydrolyzed in aqueous phase. Oxidation is a process that involves the transfer of electrons between atoms. This brings out the reaction of combination and separation of ions or molecules. Redox potential indicates the oxidation state of multivalent metal ion in the aqueous environment.

5. Monitoring Well/Borehole

Groundwater contamination can be detected by using either groundwater monitoring well/borehole (direct method/sampling) or a geophysical technique (indirect method/nonsampling).[32] Geophysical techniques operate by imposing stress on the subsurface, measuring the response toward the imposed stress, and using the data to assess the nature of the groundwater body. Some examples of the geophysical method involve the use of heat conduction, magnetism, shock waves, and electricity. However, numerous studies have found that the detection of groundwater contamination using this method can be very difficult since the data can be affected by a number of factors such as contaminant type, soil type, soil porosity, soil moisture, moisture profile, and clay content.

For routine monitoring purposes, usually the groundwater-monitoring method is used. This is the same method applied at the Pulau Burung landfill site. A monitoring well or borehole is constructed at the landfill site. The diameter of a monitoring well is much larger than that of the borehole. The construction of a monitoring well is accomplished through digging, while a borehole can be constructed just by drilling. For groundwater monitoring, usually a series or network of monitoring points is constructed within the landfill site. In the case of Pulau Burung landfill site, a monitoring well was positioned at both leachate collection ponds, in the middle of the landfill and at site boundaries where the contaminated groundwater will exit the gradients. The monitoring points of the five boreholes at Pulau Burung landfill site are shown in Fig. 10.

5.1. *Type of Monitoring Well/Borehole*

Many types of monitoring wells/boreholes are used to retrieve groundwater samples. The most common type is through the construction of a conventional borehole at the monitoring point. Other monitoring types derived for sampling are either a combination of borehole and piezometer installation or other multilevel sampling devices.

5.1.1. *Open and Fully Screened Boreholes*

The uncased boreholes or fully screened boreholes have been used to sample groundwater for many years. Open/uncased boreholes are used at sites with stable soil while fully screened boreholes are used for unconsolidated soil types.[15] A fully screened method is applied for the sampling of groundwater at the Pulau Burung landfill site. The sample is obtained by pumping it from a depth or a number of samples taken from the boreholes. This method provides a depth-averaged sample of groundwater since the pumping would force the water from the surrounding groundwater to enter

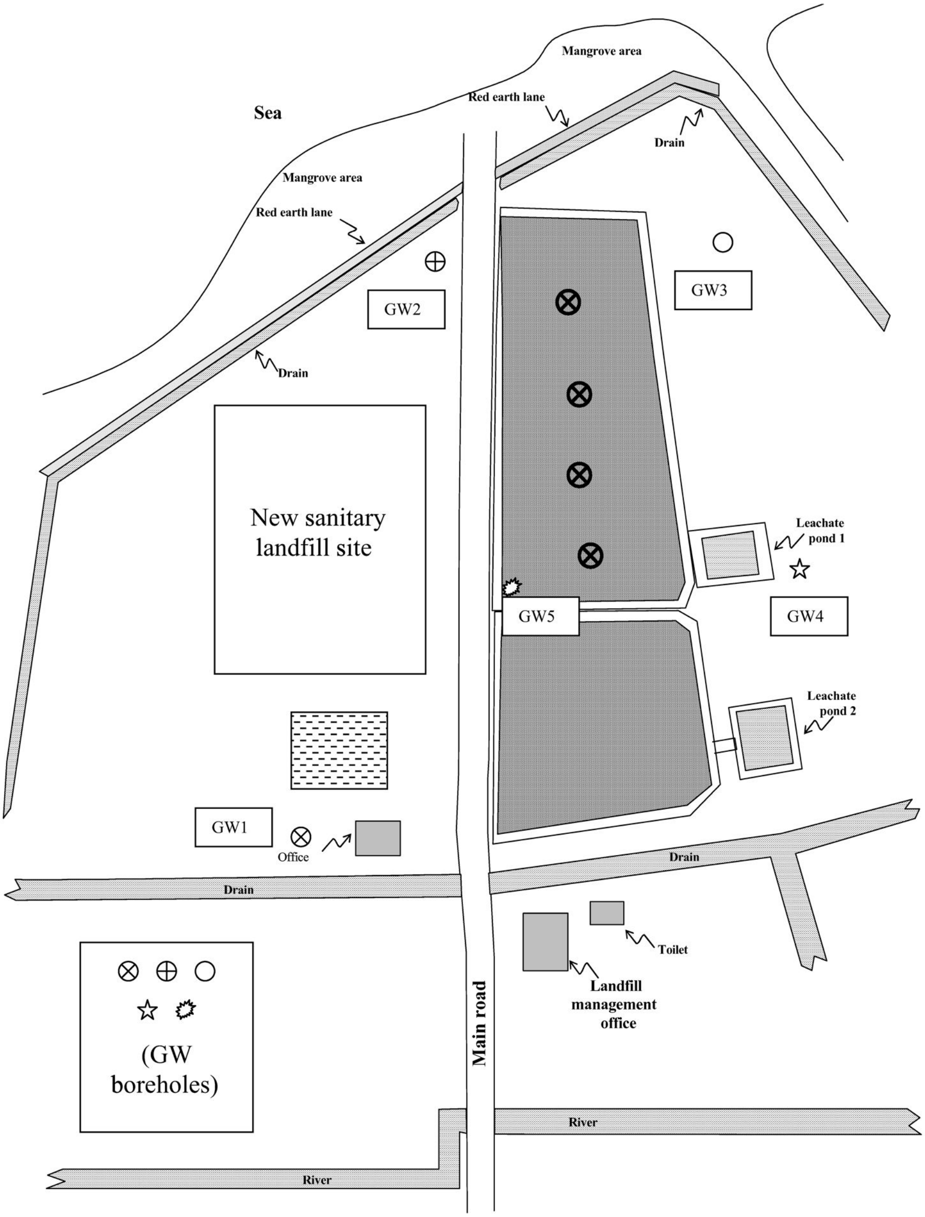


Figure 10. The Monitoring Points of the Five Boreholes at Pulau Burung Landfill Site.[9]

the casing of the borehole. However, prepumping or purging to remove the stagnant water in the borehole casing is necessary as the step will draw the fresh groundwater from the surrounding aquifer to be taken as samples. Sampling without prepumping or purging would result in an error of the sampling procedure and will be discussed

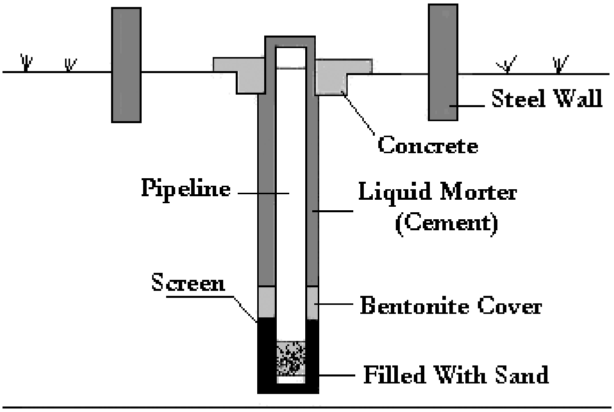


Figure 11. Schematic Design of an Open Borehole at Pulau Burung Landfill Site.[9]

further in this chapter. Figure 11 shows an example of a fully screened borehole built at Pulau Burung landfill site.

5.1.2. *Straddle Packer and Multipacker Boreholes*

This method is similar to the open or fully screened boreholes. The difference is that this method uses packers to restrict the vertical movement of water during the sampling. Using packers ensures that the pumping would only suck groundwater from a certain section of an aquifer since it is being isolated by the packer.[19] However, the difficulty in using this method is normally caused by the irregularity of the borehole wall and results in operational problems. Research by Barczewski and Marschall (1989) concluded that this method would give the same depth-averaged samples as the open or fully screened borehole method.

5.1.3. *Short-screened Boreholes and Piezometers*

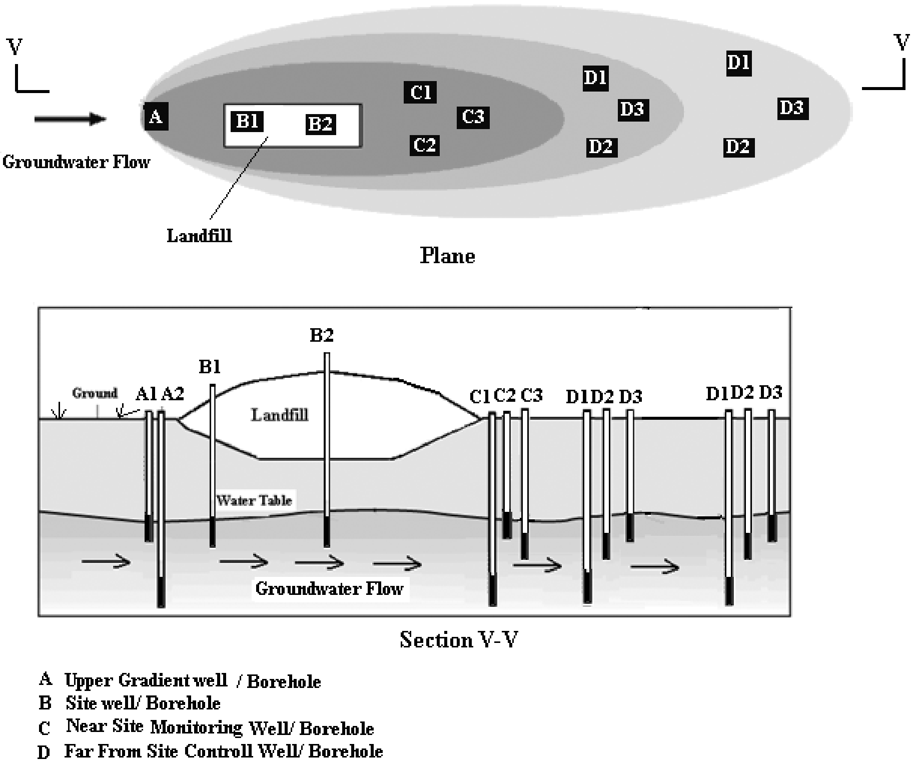
With this method, a series of short-screened boreholes, with 1–2-m-long screens are set on different depths and separated by low-permeability bentonite seal. One or a bundle of nested piezometers can be placed within the borehole for abstraction of groundwater sample. This method is quite effective since it only requires the use of small diameter pump and only a small amount of stagnant groundwater within the screen needs to be excreted. In order to obtain a representative sample, purging of at least 2–3 borehole volumes is recommended.[20] It is a cost-effective method for determining the variability of groundwater quality at different depths. However, it is necessary to reduce the rate of pumping for low-permeability soil since it could damage the pump if the boreholes are emptied during the sampling.

5.1.4. *Multilevel Boreholes and Lysimeter*

This method is designed to simplify the sampling procedure by eliminating the need to remove stagnant water and reducing the screen length and tube diameter. This sampling method usually uses gas lift or suction to remove water from the sampling device to obtain the sample.

5.2. *Coordination, Number, and Depth of a Borehole*

The coordination of monitoring wells is determined by assessing the risk of contamination at the landfill site. Selection of the monitoring position is made by determining the pathway in which contamination would likely to occur; it should be as close as possible to the leachate source. The number of wells chosen also depends on the risk assessment of the landfill site. Usually, at least three monitoring wells are required in which one is located on the upper gradient of the landfill site and the other two at the lower gradient. Figure 12 shows the groundwater-monitoring wells based on the upper gradient and lower gradient flows at landfill site.



Plane

Section V-V

Figure 12. Groundwater-monitoring Well Based on the Upper-gradient and Lower-gradient Flows.[10]

In the case of the Pulau Burung landfill site, monitoring is selected based on the location where infiltration of leachate is likely to occur. Additional monitoring wells are also installed at the boundaries of the landfill site to provide baseline data for future extension of the landfill area. These additional wells also help determine the risk of natural water resource pollution surrounding the area. The numbers of additional wells constructed is also determined by the type of liner used at the site. At least one additional monitoring well is required per 100 m width of landfill site for a landfill site with a natural liner and per 50 m width for an artificial liner. This ratio is due to the probability of leakages by diffusion for landfill site with natural liner, and leakages through hole or tears for the artificial liner. If leachate contamination is discovered, other monitoring wells are usually needed, including a well outside the landfill area. The depth of a monitoring well/borehole depends on the regional water table and the coordination of monitoring point in the landfill site, which is either on natural ground or on the waste layer as shown in Fig. 12.

5.3. *Construction of a Well/Borehole*

A well or borehole can be constructed either during landfilling or retrofitted after landfilling. Both methods of monitoring design have their own advantages and disadvantages as summarized in Table 10. Drilling of a borehole for groundwater sampling must be done with precautions in order to avoid contamination or damage to the liner system, especially in the case of a monitoring point located on the waste

Table 10. Monitoring Design on Landfill Site.[1]

Type of Monitoring Installation	Advantages	Disadvantages
Retrofitted	Made vertically Monitoring well with higher density can be built where required Easier selection for design and sealing	Difficult due to risk of puncture Unpredictable drilling problem Potentially hazardous during drilling stage Usually required large or more specialize drilling rig for over 30 m deep
Built	Monitoring from base of landfill drainage layer Install on site Obtain monitoring data during landfill operational	Prune to damage during landfill operation Design might be effected or disintegrated by chemical Required additional foundation system for maintaining vertically and prevent puncture No capping and restoration

Table 11. Type of Drilling Method for Construction of Borehole.[1]

Drilling Rig	Advantage	Disadvantage
Rotary rig	Inexpensive; Quick and easy cleaning; Lining can be put directly into the hollow; Required no drilling liquid; Allow continuous sampling in consolidated material	Cannot drill through hard rock; Depth and water strike cannot be easily identified using solid stem auger; Solid stem auger cannot be used on loose ground; Hollow stem auger cannot drill through presence of boulder; Installation of annular fill and seal cannot be done in easily collapsed ground
Cable rig	Inexpensive; Easy to clean and to identify changes in water; Use little amount of drilling fluid; Possible to obtain undisturbed bulk sample; Required temporary casing for installation of lining but with more accuracy	Cannot drill through hard rock; Slow; Can damage the side of borehole
Other rotary rig	Could be inexpensive and fast in unconcealed material; Continuous sampling can be made in consolidated material; Can be use for drilling in all formation type	Can be expensive; May required drilling fluid and possibility of contaminant due to circulating fluid; Can damage the side of borehole; Sampling can be slow at great depth; In unconsolidated formation, only installation of narrow diameter lining permitted

layer. Common drilling methods on the waste layer include continuous flight augers, hollow stem augers, single flight augers, large diameter single-tube barrel, and cable percussion (shell and auger). For natural ground drilling, common drilling methods are auger (hollow stem augers, continuous flight, or single flight), cable percussion (shell and auger), and conventional rotary. Table 11 summarizes the different methods of drilling and their advantage and disadvantages.

5.3.1. *Contamination During Drilling*

Drilling of a borehole must also be done properly in order to avoid contamination of groundwater during the process and to avoid damage to the constructed borehole. Contamination may originate either from the addition of drilling fluid or contaminated drilling equipment. The addition of drilling fluid or water is unavoidable; therefore, water from a known quality source is normally used and the characterization of the added water can be provided as a reference for the groundwater data. Meanwhile, to prevent contamination due to contaminated drilling equipment, decontamination of the equipment is necessary after each drilling exercise, especially if drilling is done in a contaminated ground. Decontamination of drilling parts is done by pressure-washing, rinsing, or steam cleaning.

5.4. *Borehole Maintenance and Cleaning*

A borehole must be maintained in order to prevent silt accumulation and blockage by other objects. Therefore, the cleaning of a well/borehole is required to remove silt and other material from the lining, gravel pack, and strata. Cleaning is done by surging and pumping the borehole for a certain period. The volume of water used for cleaning should exceed the normal borehole volume (pre pumping or purging) to achieve adequate cleaning. A lesser volume may be used if a geotextile wrap is used for the borehole design. Maintenance is not only needed for active boreholes but also for abandoned and damaged boreholes. These boreholes must be sealed and capped with either cement or bentonite to prevent contamination of groundwater through the borehole passage.

6. Sampling and Measurement

6.1. *Measurements*

Measurements taken at a landfill site should include physical measurements, principle chemical composition, minor chemical composition, and biological measurement. For example, physical measurements would include landfill site runoff, vegetation presence, rainfall measurement, surface water flow rate, both leachate and groundwater level, and the base of a monitoring point level. Table 12 lists the parameters for these types of measurements. As in the case of sampling for other water sources, sampling at wells/boreholes can be done either as a discrete or spot sample, or composite or continuous sample. Discrete or spot samples are taken at a single point, whereas composite samples are taken from a number of monitoring points or at different time intervals. Data recorded by data loggers and electronic instrumentation are called continuous samples.

6.2. *Sampling and Measurement Devices*

6.2.1. *Level Measurement*

Level measurement is used to measure the water level inside a borehole. The most common device used is an electric tape, which uses a probe submerged in the water

Table 12. Parameter for Physical, Chemical, and Biological Measurements.[1]

Measurements	Parameters
Physical	Water balance (rainfall and meteorological data and volume changes in landfill site); Observation (surface run off, vermin, vegetation, and other contaminants source); Groundwater and leachate level
Principle chemical composition	pH, temperature, EC, DO, Eh, TSS, TDS, NH_3-N, TON, VFA, TOC, BOD, COD, Ca, Mg, Na, K, total alkalinity, SO_4, Cl, Fe, Mn
Minor chemical composition	Trace metals, inorganic (Cd, Cr, Cu, Ni, Pb, Zn, orthophosphate, As, Ba, B, CN, F, Hg, dissolved methane), and organic (phenols, mineral oils, pesticide, polychlorinated biphenyl, and chlorinated solvents) contaminants
Biological	Coliform bacteria, chlorophyll a, toxicity test, macroinvertebrate community

to form an electrical circuit. Inaccurate results from the device may occur in different conductivity waters (high or low) and in different environments. To overcome these errors, a sensitivity switch can be used and regularly calibrated for the dipping purpose. Other devices used for this measurement are the float device or pressure transducer, which are used for remote or continuous monitoring, together with a data logger or chart recorder. A pressure transducer operates on the pressure of the groundwater level and further depths would result in much higher pressure. Therefore, a range of depth must be determined and calibrated to obtain accurate and reliable data.

6.2.2. *Sampling Apparatus*

At the Pulau Burung landfill site, groundwater samplings were obtained through the use of a suction pump called the Masterflex Composite Sampler. Examples of sampling done at the landfill site are shown in Figs. 13 and 14. Besides the suction pump, many types of sampling devices are used for the groundwater borehole as summarized in Table 13.

6.3. *Sampling Procedures*

Before groundwater abstracted from a borehole can be analyzed, prepumping or purging of stagnant water within the borehole is necessary. This is because the stagnant water that remains in the borehole no longer has the same characteristics as the surrounding water (groundwater). The stagnant water might have undergone significant chemical changes due to long contact with atmospheric pressure degassing, contamination, and biological processes. Usually three borehole volumes (internal radius of borehole × height of water column in borehole) are purged before sampling is done. However, purging also creates some problems since a large volume of water

Figure 13. Abstraction of Groundwater Sample at GW2 Point.[9]

Figure 14. Abstraction of Groundwater Sample at GW5 Point.[10]

needs to be discarded and will also draw fine matters that introduce high suspended solids into the sample. Filtration of a sample can be done to resolve the problem of suspended solids. Filtration may be conducted during sampling or in the laboratory.

While samples are typically analyzed in the laboratory, some parameters must be measured on site such as pH, temperature, electrical conductivity, dissolved oxygen, and redox potential. These parameters must be determined on site since

Table 13. Various Sampling Devices.[1,2]

Equipment	Advantage	Disadvantage
Bailer	Easy to operate and portable; Low cost; Dedicated or disposable	Slow purging due to low abstraction; Sampling the top of water column; Bailing cable can cause cross-contamination due to movement and agitation if used vigorously
Discrete depth sampler	Easy to operate and portable; Low cost; Sampling different depth by sequential sampling	Slow purging due to low abstraction; Failure in closure due to presence of suspended solid; Causes agitation and thus causes pressure change
Inertial	Can be used for purging; Can operate to up to 60 m depth and in silty condition; Low-cost dedicated system; Simple maintenance; Available in lightweight and portable	Causes agitation; Will entrain the suspended solid in the sample
Suction (peristaltic)	Tubing can be installed in the hole; Avoid cross-contamination by the used of inertial pump as priming mechanism	Suction can degas sample; Causes agitation and thus causes pressure change; Only operates up to the depth of 7.6 m or less; May cause cross-contamination thus requiring priming
Electric submersible	Easy to operate and can be used for purging even for low flow purging; Operates deeper than 75 m using pump with diameter more than or equal to 50 mm	Causes agitation and thus causes pressure change; Heavy and requires vehicle for transport; Capability reduced in higher temperatures and presence of suspended solids
Bladder	Can be used for low purging; Operates at any depth; Small disturbance to sample	Slow purging due to low abstraction; Expensive;
Gas lift	Operates at any depth	Requires compressor or tank; Degassed and pressure change of sample due to usage of gas

Table 14. Recommended Storage and Preservation of Groundwater Sample.[21]

Parameter	Recommended Container	Max Holding Time	Preservation Required
Temperature	Polyethylene, glass	Analyze immediately	None
pH	Polyethylene, glass	Analyze immediately	None
Alkalinity	Polyethylene, glass	48 h	4°C
BOD	Polyethylene, glass	24 h	4°C
COD	Polyethylene, glass	48 h; 28 days if acidified	4°C; acidify with H_2SO_4 to pH <2, 4°C
VOC	Glass, teflon	14 days	4°C; acidify with HCl to pH <2, 4°C
TDS	Polyethylene, glass	7 days	4°C
Ammonia	Polyethylene, glass	7 days; 28 days if acidified	4°C; acidify with H_2SO_4 to pH <2, 4°C
Nitrate	Polyethylene, glass	48 h	4°C
Heavy metals (include iron and manganese)	Polyethylene, glass	6 months except 28 days for Hg	Acidify with HNO_3 to pH <2, 4°C (for dissolved metal, filter before acidify)
Calcium, magnesium, sodium, potassium, fluoride, sulfate, chloride, hardness	Polyethylene, glass	28 days	4°C

significant changes of groundwater may occur as soon as it comes in contact with the atmosphere. To prevent contamination of a sample, it is advisable to analyze the sample within 24 h, under limited exposure to sunlight, stored at low temperatures, and without agitation of the sample. For lab analysis, an appropriate container is needed. Certain analyses of parameters will also require the preservation of a sample as summarized in Table 14.

Frequencies of sampling can be determined using the groundwater flow rate calculated using Eq. (1). From the flow rate, distance and travel time of contaminants are calculated using:

$$t = s/Q, \tag{2}$$

where, t = travel time (s) and s = distance (m).

This equation takes into account the flow rate of groundwater; thus, it can only be used to determine the minimum groundwater frequencies since movement of contaminants also influenced by other factors such as different flows between soil pores and attenuation mechanisms.

6.4. *Sampling Error*

Sampling error must be prevented in order to obtain an accurate representative data. Certain factors must be considered especially during the sampling of groundwater. These factors include the sampling procedure and the selection of sampling devices. Table 15 summarizes the factors that affect the quality of groundwater sample and the affected parameter.

Table 15. Error in Sampling and Affected Parameter.[1]

Process	Contaminant Affected	Causes	Prevention
Inappropriate sampling	Alkalinity, BOD, COD, DO, TOC, VOC, NH_3-N, metal, phosphate, organic compound, and dissolved gases	Still water in the well are not purged before sampling	Appropriate selection of purging procedure
Pressure change	Alkalinity, DO, VOC, NH_3-N, and dissolved gases	Sampling method — movement of equipment causes small pressure change	Reduce or eliminate movement part of equipment during sampling
	DO, VOC, and dissolved gases	Change of pressure — underground to ambient pressure	Sustain pressure if possible
Temperature change	DO, VOC and dissolved gases	Sample storage	Do analysis immediately
Aeration/ oxidation	DO, VOC, NH_3-N, trace metal, and dissolved gases	Contact with air causes release of gases and volatile compounds and the precipitation of metals	Reduce contact with air and use air-tight sampling bottle
Adsorption/ dissolution of metals	Trace metals	Silt in water sample — problem with some trace metals such as zinc, iron, and manganese	Prevent suction of silt together with water sample
Cross-contamination	DO, VOC, trace metals, trace organic compounds, and dissolved gases	Sample equipment and handling	Equipment for leachate and water samples should be separated
Adsorption/ desorption of organics	BOD, COD, TOC, and trace organic compounds	Material in sampling equipment and borehole	Selection of appropriate tube, filter and design

7. Data Assessment

7.1. *Record*

7.1.1. *Baseline/Background Data*

In order to determine whether or not there is significant contamination in the groundwater data, another set of data is required for comparison with the data taken during or after landfilling is completed. This set of data is called baseline data or background data. There are two ways for determining baseline data for groundwater in landfill site. These are:

1. *Historical Data*: Groundwater data that has been taken on the site before the landfill started its operation.
2. *Upper Gradient Data*: Groundwater data that has been taken on the same groundwater body as the one in the landfill site but on the upper gradient of the site. This gives groundwater data that is not influenced by leachate infiltration.

However, it should be emphasized that even baseline data itself may have variations since it is still affected by other factors besides landfill, such as seasonal factors like rainfall. A set of data is required, so that the total variability of the baseline data is used for the comparisons. In the case of the Pulau Burung landfill site, it began its operation as a dumping site; therefore, it did not have any historical baseline data for comparison with the current data. However, for further expansion of the landfill site, necessary steps have been taken to install two boreholes at the potential sites to indicate the baseline data for future use.

7.1.2. *Borehole Log*

Log data for borehole installations at the landfill site are kept as records for safety and maintenance. Information that is included in the log data are well number, well location (grid point), date of installation, type of well, type and diameter of well casing, mean sea elevation of the top well casing, ground surface near well, depth of the well, top of the screen, and material and length of well screen.

7.2. *Data Analysis*

Current data and the baseline data are compared to determine whether or not the contamination of leachate occurs at the landfill site. In the case of Pulau Burung, two sets of data were compared from research done by Yusof[9] and Izham.[10] The comparison showed very significant changes in groundwater parameters during the

Table 16. Average Value of Parameter Sampled at Monitoring Wells, GW1 to GW5.[9,10]

	(01/10/2001–23/10/2003)				
Parameter	GW1	GW2	GW3	GW4	GW5
COD (mg/L)	153	116	655	585	541
pH	8.19	7.08	6.67	6.7	7.28
Turbidity (NTU)	96	12	13	109	190
SS (mg/L)	104	20	51	63	115
Color (PtCo)	376	599	301	199	330
Manganese (mg/L)	1.20	0.40	0.20	0.70	1.71
Lead (mg/L)	0.88	0.77	0.77	0.64	0.80
Mercury (mg/L)	0.01	0.01	0.01	0.01	0.01
Copper (mg/L)	0.22	0.19	0.26	0.21	0.51
Total chromium (mg/L)	0.10	0.10	0.12	0.11	0.09
Phosphate (mg/L)	0.3	0.3	0.3	0.7	1.3
Sulfate(mg/L)	18.7	4.2	0.7	0.9	2.1
Nitrate (mg/L)	5.7	2.5	1.8	3.6	7.9
BOD (mg/L)	23	19	160	120	135
Temperature (°C)	25	25	31	30	31

one-year time frame as shown in Table 16. An increase in certain parameters was discovered especially for BOD, COD, turbidity and SS.

7.3. *Data Representation*

As data are recorded for a number of monitoring points at certain time intervals, it would be effective to portray such data in a graphical format such as a time series chart or spatial plot. An example of groundwater-monitoring data obtained from Pulau Burung landfill site over time is shown in Figs. 15 and 16.

Acknowledgments

The authors acknowledge the Ministry of Science, Technology and Environment Malaysia for the National Scientific Fellowship and Intensification of Research in Priority Area (IRPA) research grant provided by the Ministry this chapter. The authors also wish to acknowledge the cooperation of Majlis Perbandaran Seberang Perai, Penang and Idaman Bersih Sdn. Bhd., Penang during the study.

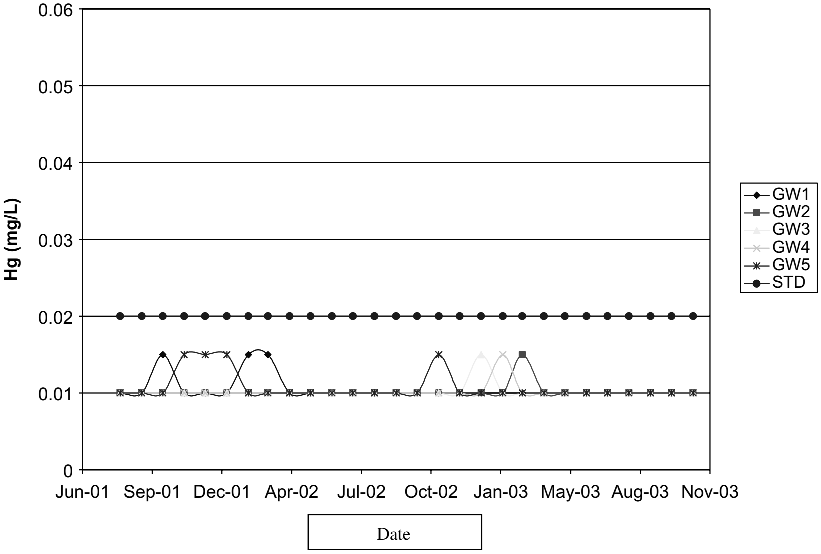


Figure 15. Time Series Graph for Hg (August 8, 2001–November 5, 2003).[10]

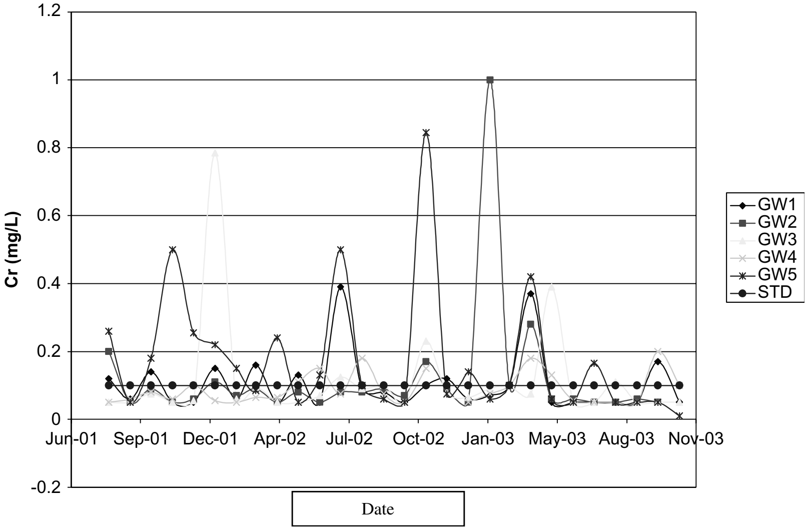


Figure 16. Time Series Graph for Total Cr (August 8, 2001–November 5, 2003).[18]

Nomenclature

BOD	Biochemical oxygen demand, mg/L
COD	Chemical oxygen demand, mg/L
DO	Dissolved oxygen, mg/L

EC	Electrical conductivity, μS/cm
Eh	Redox potential, mV
i	Hydraulic gradient, m/m
K	Hydraulic conductivity, m/s
n	Effective porosity, dimensionless
NH_4-N	Ammoniacal nitrogen, mg/L
PBLS	Pulau Burung landfill site
Q	Flow rate, m/s
s	Distance, m
SO_4	Sulfate, mg/L
t	Time, s
TDS	Total dissolved solids, mg/L
TOC	Total organic carbon, mg/L
TON	Total oxidized nitrogen, mg/L
TSS	Total suspended solids, mg/L

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Chapter 14

REMEDIATION OF CONTAMINATED GROUNDWATER

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Abstract

This chapter provides an overview of groundwater contamination problems, common types of groundwater contaminants, contaminant transport processes, groundwater cleanup goal and strategy, and remedial technologies. The remedial technologies covered are containment method, pump-and-treat, surfactant/cosolvent flushing, air sparging, permeable reactive barrier, bioremediation, and a few other *in-situ* cleanup technologies. The system installation and design, performance monitoring, and advantages and limitations associated with each of the cleanup technologies are discussed.

Keywords: Groundwater contamination, *in-situ* cleanup, cleanup strategy, monitoring, contaminant transport.

1. Introduction to Groundwater Contamination

Groundwater contamination occurs when hazardous chemicals or waste leachates come into contact with groundwater and make the water unsafe for human use. Even though groundwater contamination may be defined by law and regulation, regulatory definitions for acceptable concentrations of specific contaminants may differ between nations or provinces or between national and provincial/state levels, with the differences generally being based on different judgments as to the degree of risk posed by the contaminants.

The main anthropogenic sources of contaminants are various industries that emit hazardous chemicals. Contaminants are released and enter the subsurface systems either directly by spill, leakage of storage tanks and conduits, land disposal, injection, surface application or indirectly through atmospheric deposition. There are a great variety of waste materials produced by various industrial or commercial activities that contribute to the sources of contaminants. Table 1 lists the major industrial sources and the wastes or contaminants produced. These contaminants are released into the subsurface systems in solid, liquid, or gaseous form.

Another major source of groundwater contaminants is associated with agricultural activities. In agricultural practice, pesticides, fertilizers, compost, and

Table 1. Industrial Sources and Types of Contaminants or Wastes Produced.

Sources	Types of Contaminants
Gasworks	Aromatic hydrocarbons, phenols, and CN^-
Waste landfill sites	All kinds of organic and inorganic contaminants
Chemical production and handling sites	All kinds of organic and inorganic contaminants (except no spent plating wastes)
Metal plating and cleaning industries	Heavy metals, CN^-, organic solvents, spent plating wastes, acids, and bases
Metal smelting	Heavy metals, metal slag, smelting sludges, and acids, oils
Pesticide manufacturing sites	Pesticides, solvents, Hg, As, and Cu
Automobile service facilities	Pb and hydrocarbons
Laboratories	Acids, bases, heavy metals, inorganics, ignitable wastes, and solvents (all types of chemicals are possible)
Printing plants	Acids, bases, heavy metals, inorganics, ink, solvents, spent plating, and sludges
Mining sites	Acid mine drainage ($2 < pH < 4$, rich in H_2SO_4), waste waters containing various heavy metals and organic solvents, and tailings that contain heavy metals and “unwanted” minerals or slag
Petroleum refineries	A wide range of hydrocarbons

sometimes animal wastes, human wastes, or sewage sludge are applied to the cropland. As a result, pesticides, heavy metals, and toxic organic compounds tend to accumulate in the soil, which may eventually enter groundwater. Excessive use of fertilizers also makes nitrate the most ubiquitous contaminant in groundwater. Animal and human wastes contribute to the elevation of nitrate, phosphorous, pathogens, and viruses in groundwater.

Based on a geographical scale, groundwater contamination sources can be classified as point sources and nonpoint sources. Point sources are those that release contaminants from a discrete geographic location, including leaking underground storage tanks (USTs), ruptured transfer pipes, septic systems, injection wells, and spillage due to traffic accidents. Nonpoint sources of contamination are more extensive in area and diffuse in nature. Agricultural runoff (which carries pesticides and fertilizers), urban runoff, and atmospheric deposition are the potential nonpoint sources.

Groundwater contamination can occur through various processes. Water that precipitates onto the earth and slowly infiltrates into a contaminated soil surface, waste piles, or waste landfill sites dissolves materials with which it comes into contact, forming a leachate that may contain inorganic and organic contaminants. The leachate will continue to migrate downward under the influence of gravity until it reaches the saturated zone, contaminating the shallow groundwater aquifer. Storage sites and landfills excavated to the saturated zone may also permit direct contact of contaminants with groundwater. When a polluted surface water body recharges into uncontaminated groundwater, it will contaminate the groundwater. Leaked USTs and pipelines that lie within the saturated zone can contaminate groundwater directly. In another scenario, contaminants enter the aquifer system from the surface by leakage through injection wells because of waste disposal through the deteriorated or improperly constructed wells. In stratified geological formation, contaminated groundwater can mix with uncontaminated groundwater through a process known as inter-aquifer exchange, in which one water-bearing unit communicates hydraulically with another. This occurs most commonly in bedrock aquifers where a well penetrates more than one water-bearing formations to increase its yield.[1]

Groundwater contamination has been increasingly threatening the world's already scarce freshwater resources. Although many groundwater restoration programs have been implemented in decades, the problem is worsening due to continual contamination by negligence and illegal activities, and ineffective remedial systems implemented early on. However, valuable experiences have been gained from the past and several innovative remedial technologies and effective remedial strategies have been developed over the past decades. To implement an effective groundwater cleanup, one should begin with a correct understanding of various aspects of the sites. The following issues must be understood:

- Types and behaviors of groundwater contaminants,
- Transport and distributions of contaminant,
- Characteristics of the source zone,
- Limitations and complexity of groundwater cleanup,
- Site hydrogeology, existing facilities, and site constraints,
- Importance of cleanup strategy,
- Cleanup goal, and
- Cleanup technologies and selection.

2. Types and Behaviors of Groundwater Contaminants

A large number of contaminants are found in contaminated groundwaters and may be classified as follows:

- Metals,
- Inorganic nonmetals,
- Petroleum hydrocarbons,
- Nonhalogentaed solvents,
- Halogenated solvents,
- Pesticides,
- Oxygenated compounds,
- Polychlorinated biphenyls (PCBs),
- Polycyclic aromatic hydrocarbons (PAHs),
- Dioxin and furans,
- Industrial intermediates,
- Drugs and pharmaceuticals, and
- Radionuclides.

Understanding the behavior of contaminants is important for contaminated groundwater management, risk assessment, and planning and designing a cost-effective remedial system.

2.1. *Organic Contaminants*

Most organic contaminants enter the subsurface as constituents of a liquid phase. The liquid phase may be a concentrated leachate, a dilute aqueous solution, or an organic liquid that is immiscible with water (or commonly referred to as a nonaqueous phase

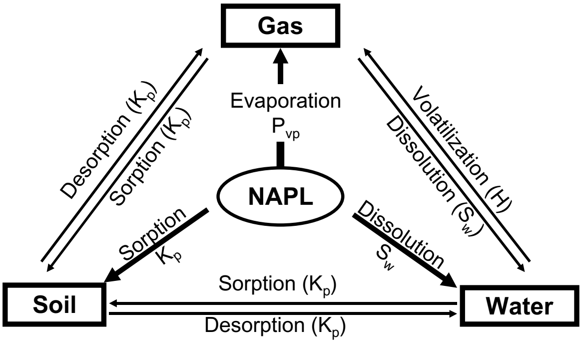

Figure 1. Four-Phase Diagram Showing the Distribution of an Organic Compound Among Various Phases.

liquid (NAPL)). Characteristics of the "carrier" fluid can greatly influence the fate, transport, and persistence of a specific compound in the fluid.

Organic contaminants present in subsurface may show up in the following phases:

- Solid phase (adsorbed onto soil solid constituents or soil biota),
- Vapor phase (in the vadose zone),
- Dissolved phase (in contaminated groundwater and vadose-zone pore water), and
- NAPL.

Figure 1 shows a four-phase diagram and the processes that affect the distribution of an organic compound among these phases. The distribution depends on physicochemical properties of the compound, such as vapor pressure (P_{vp}), Henry's law constant (H), partitioning coefficient (K_p), and solubility in water (S_w). The critical molecular property of an organic contaminant that affects these properties is molecular polarity. Polar compounds are hydrophilic while those nonpolar are hydrophobic.

Several processes affect the concentration, physicochemical form, transport rate, degradation, and ease of removal of organic substances from the subsurface. These processes include physicochemical processes, biotic transformation, and abiotic transformation. The physicochemical processes are dissolution, adsorption/desorption, volatilization, immiscible phase separation, cosolvation, and ionization. Biotic reactions involve metabolically active organisms, products of living cells, or products of dead organisms, while abiotic reactions involve none of these.

2.2. *Inorganic Contaminants*

Inorganic contaminants of major concern are heavy metals or trace metals. The most frequently found harmful metals in the contaminated groundwater are arsenic (As), chromium (Cr), lead (Pb), zinc (Zn), cadmium (Cd), manganese (Mn), copper (Cu), barium (Ba), and nickel (Ni).[2] Nonmetal inorganic contaminant of public health concern recently are selenium (Se), perchlorate, and nitrate. Selenium has been identified as the cause of poisonings and deaths of wild birds at the Kesterson Natural Wildlife Refuge in California in the early 1980s.

Not all metals present in the subsurface are mobile or bioavailable to plants or soil microorganisms and invertebrates. A metal in groundwater usually shows up in the dissolved and solid phases. However, the microbial-mediated methylation process can transform some toxic elements such as Hg, As, Pb, and Se into volatile organometals and, therefore, increase their mobility. Only the mobile fraction of metals is of environmental significance, as it has the potential to reach receptors through various pathways. The hazard posed by inorganic contaminants also depends on the oxidation state in which they are present. Cr(VI) and As(III) are very toxic and mobile compared to Cr(III) and As(V), respectively. Se(IV) is less mobile but more toxic than Se(VI).

Physical and chemical interactions between metals and soils are the most significant reactions that influence the mobility of the metals in the subsurface. Examples of the physical interactions are intraaggregate and intraparticle diffusions while the chemical interactions are adsorption/desorption, precipitation/co-precipitation/dissolution, complexation, and oxidation/reduction. Factors affecting these interactions or processes are:

- Soil composition (soil organic matter (SOM), clay, oxides, etc.) and properties (cation exchange capacity (CEC), alkalinity, porosity, etc.),
- Characteristic of the metals and their speciation,
- Hydrogeologic characteristic, and
- Geochemical parameters of groundwater, including pH, alkalinity, redox condition, ionic strength, aqueous phase composition (the presence of ligands, humic substances, etc.), the presence of competitive adsorption in multicomponent system, and temperature.

3. Contaminant Transport

3.1. *Contaminant Plume Behavior and Transport*

The tendency for a particular contaminant to be transported in the aqueous phase depends primarily on its aqueous solubility. Solubilities of organic contaminants vary widely and are influenced by hydrophobicity, temperature, salinity, natural organic matter, or the presence of multiple organic compounds in solution or cosolvency.[2] On the other hand, the solubility of metals depends primarily on pH, redox potential, ionic strength, the presence of complexing ligands and dissolved organic matter, and the interactions between the metals and soil materials. Once a compound dissolves in groundwater, it will migrate in the direction of groundwater flow and will create a contaminant plume in aquifer.

As the flow rates of natural groundwater are slow in most geologic formations, a contaminant plume usually migrates very slowly, for example, at a rate of one to thousands of meters per year. The rate of contaminant plume movement largely depends on the type of geologic material through which it is moving. More rapid movement can be expected through coarse materials such as sand or gravel than through fine-textured materials such as silt and clay. In addition, the presence of fractures will provide an easy path for faster plume migration. The physical and chemical compositions of the geologic material are equally important too. Fine-textured materials with high clay and organic matter content tend to impede contaminant migration by having a low hydraulic conductivity and through sorption of contaminants onto the reactive soil materials. Sorption, precipitation, volatilization, and degradation can retard contaminant movement in the aquifer and lead to attenuation of a groundwater contaminant plume.

Transport processes such as advection, mechanical dispersion and diffusion, hydrogeologic properties, and contaminant characteristics determine the shape and migration of contaminant plumes. Although advection and mechanical dispersion play the primary role in the contaminant plume migration, dissolved contaminants also move along concentration gradient through diffusion. While diffusion is an extremely slow process in water, over large time scales, it can result in the transfer of contaminants to less permeable zones in the aquifer. Both dispersion and diffusion, collectively known as hydrodynamic dispersion, tend to spread the contaminant plume in directions parallel and perpendicular to the groundwater flow direction. The spreading can be enlarged by the presence of macroscopic heterogeneity in the aquifer.

The shape and movement of a contaminant plume are also dependent on the characteristics of the contaminant source, such as type of source, source mechanism, and temporal features. When contamination comes from a point source that continuously discharges contaminants, the concentrations of the contaminants are typically highest near the source. In contrast, the nonpoint sources of contamination, such as infiltration from agricultural lands, tend to result in relatively low concentrations of contaminants distributed over a relatively large area. In this situation, the boundaries of plumes are less easily defined. Temporal characteristics include the manner in which a contaminant is released over time (e.g., instantaneous versus continuous) and the time elapsed since the release. Figure 2 shows changes in plumes with respect to various source characteristics. Plume enlargement results from an increase in the rate of waste discharge to the groundwater system. Similar effects

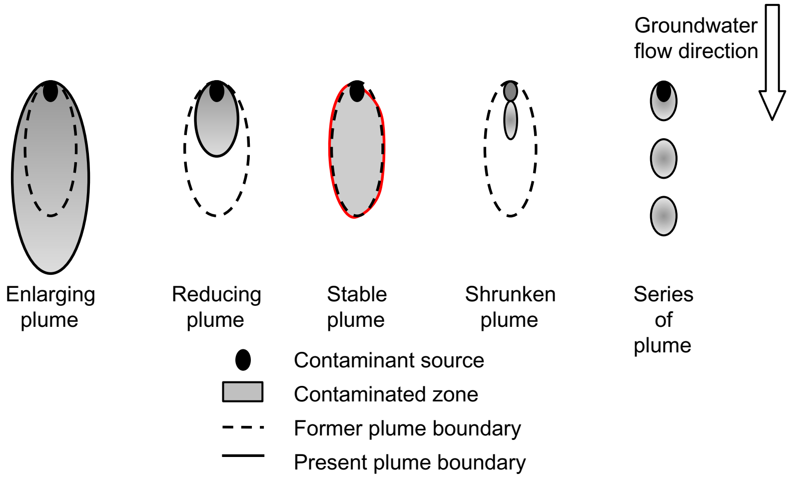

Figure 2. Possible Changes in Contaminant Plumes Due to Changes in Contamination Sources.[1]

can be produced if the retardation capacity of the geologic materials is exceeded, or if the water table rises closer to the source, causing an increase in the solute concentration.[1] Decreases in waste discharge, lowering of the water table, retardation through sorption, and reduction in groundwater flow rate can diminish the size of the plume. A stable plume suggests that the rate of waste discharge occurs at a steady state with respect to contaminant retardation and transformation processes. A plume will shrink in size when contaminants are no longer released to the groundwater system and a mechanism to reduce contaminant concentrations is present (e.g., biodegradation). An intermittent or seasonal source can produce a series of plumes that is separated by the advection of groundwater during periods of no contaminant discharge. In conclusion, hydrogeologic factors, geochemical and biogeochemical processes, contaminant characteristics, and contaminant source characteristics can affect the contaminant plume transport and distribution in the aquifer.

3.2. *NAPL Behavior and Transport*

NAPL exists as a separate phase of pure compounds in an aqueous environment, i.e., they are immiscible with water. Many NAPLs are volatile organic compounds (VOCs) of environmental concern. Examples include gasoline, kerosene, organic solvents, pesticides, PCBs, and creosote. NAPLs are regarded as a source of groundwater contamination and have been the focus of substantial research and regulatory concern because their presence requires complicated groundwater cleanup system at many hazardous waste sites.[2] Contamination by NAPLs may occur through surface spills, leaking waste lagoons, and leaking drums, pipelines, and storage tanks.

The transport of constituents in NAPLs in a subsurface involves three important mechanisms, namely, transport as a free product, aqueous transport within contaminant plumes containing the dissolved constituents, and vapor-phase transport (for volatile NAPLs).

3.2.1. *Factors Affecting NAPL Transport*

The most important hydrogeologic properties that affect NAPL transport in a subsurface are water table and seasonal fluctuations, soil stratification, soil hydraulic conductivity, degree of water saturation of pore, and pore size distribution. Fluid properties such as density, viscosity, and interfacial tension also influence NAPL behavior and transport in a subsurface. NAPLs can be classified as light NAPLs (LNAPLs) and dense NAPLs (DNAPLs), based on their relative density compared to water. When encountered by groundwater, LNAPLs tend to spread laterally at the top of the water table (Fig. 3), whereas DNAPLs tend to sink vertically (Fig. 4) and likely to fill fractures or voids in the aquitard. Viscosity governs the NAPL's rate of migration. NAPLs with lower viscosity migrate more rapidly than viscous NAPLs. When a denser and less viscous fluid displaces another of lower density and higher viscosity, narrow, preferential pathways of migration for the denser fluid known as "fingers" propagate at the displacement front. Fingering can enhance the rate of DNAPL downward migration. Interfacial tension induces the capillary forces

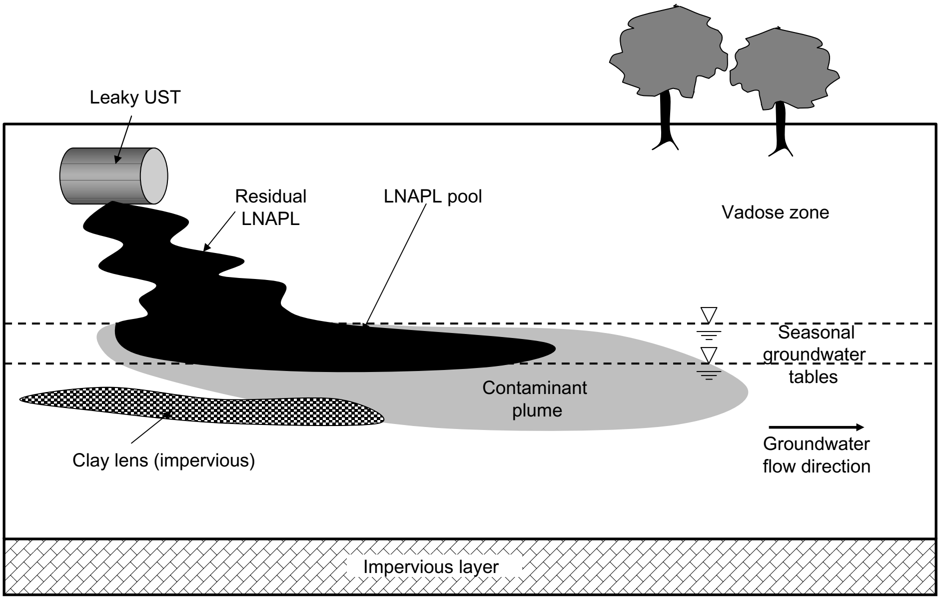

Figure 3. Conceptual Diagram of LNAPL Migration Through the Subsurface.

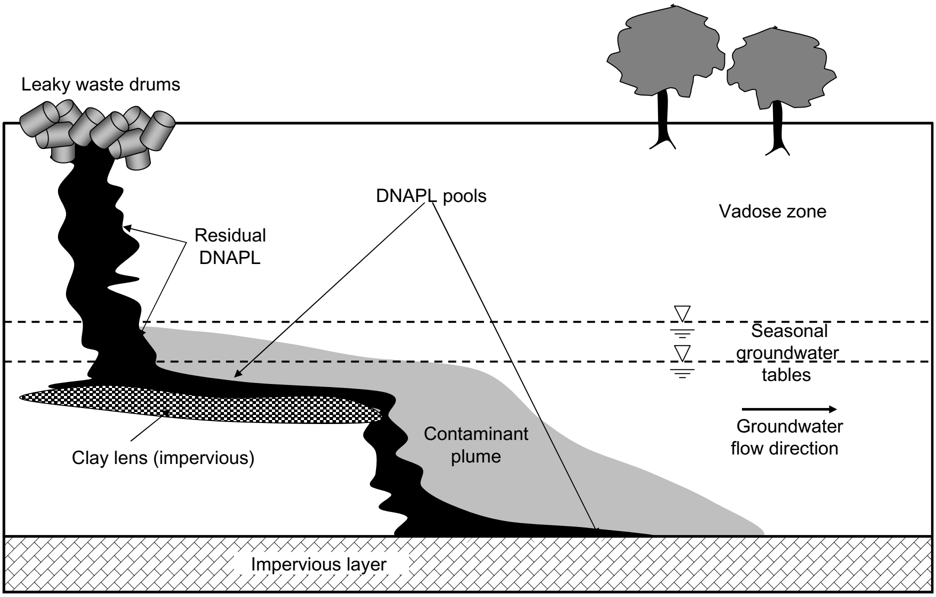

Figure 4. Conceptual Diagram of DNAPL Migration Through the Subsurface.

that determine the extent of lateral spreading of an NAPL as it migrates downward. NAPL-water interfacial tensions are typically in the order of 20–50 dynes/cm.[3] The extent and thickness of an LNAPL lens that forms at the water table decrease as the interfacial tension increases and vice versa. In summary, the pathways of NAPL migration are complex and subject to many influences. The resulting NAPL distribution is highly nonuniform and complex. These complexities hinder the task of locating and removing the contamination. Often, the presence of NAPL will remain a huge source of contaminant over a long period.

3.2.2. *NAPL Entrapment*

When NAPLs move through the subsurface, their presence causes any existing water to line the edges of the pores and adhere to soil particles, while the NAPLs tend to move through the central portions of the pores. However, a certain percentage of the NAPLs will either coat the soil particles and fill the entire pore space between soil particles or become entrapped by capillary forces in the pores forming small globules or ganglia. This entrapment occurs when capillary forces are large enough to overcome the viscous and gravitational or buoyancy forces that would otherwise displace the NAPLs. Figure 5 shows the microscopic phase distribution of NAPL in unsaturated vadose zone.

The volumetric ratio of the entrapped NAPL to the total pore volume is referred to as *saturation* (if free phase exists) or *residual saturation*. Equation (1) gives an

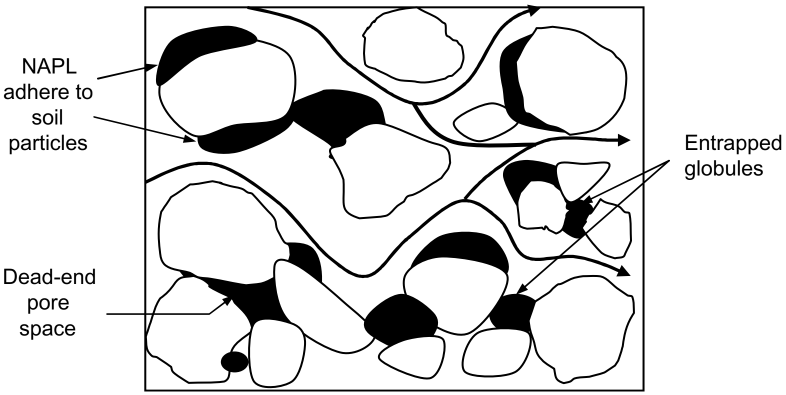

Figure 5. Conceptual Diagram of Entrapped Residual NAPL at Residual Saturation in Saturated Pore Spaces.

estimation of NAPL saturation in soil[4]:

$$S_o = \frac{\rho_b \cdot \mathrm{TPH}}{\rho_n \cdot n \cdot 10^6} \tag{1}$$

where S_o = NAPL saturation, ρ_b = soil bulk density (g/cm^3), ρ_n = NAPL density (g/cm^3), TPH = total petroleum hydrocarbon (mg/kg), and n = porosity.

The amount of saturation or residual saturation is in fact more complicated to estimate; it is a function of the initial soil moisture content, pore geometry, NAPL properties, and groundwater flow velocity. Studies have shown that in saturated media, the residual saturation values range from 15% to 50% of the total pore volume.[3,5] The value tends to be smaller in unsaturated zones due to the reduction in capillary forces in these zones.[6]

The entrapped NAPL globules cannot be mobilized by groundwater flow under typical operating conditions for pump-and-treat (PAT) systems (i.e., Darcy velocities in the range of 0.01–50 m/day) due to the strength of the capillary forces holding it in place.[7] Therefore, NAPLs present as residual saturation contamination will be physically removed only by dissolution into the groundwater or volatilization to gas phases. However, the removal of entrapped NAPL contaminants by dissolution through conventional PAT systems has been proven a very slow process, as NAPLs typically have low aqueous solubilities. In addition, water movement can be severely restricted in zones where NAPLs fill or partially fill the pore space between particles. This further hinders the removal of NAPLs by water through PAT due to restricted intimate contact area for mass transfer. Continual flushing over a long period by the moving groundwater will eventually remove most of the NAPLs but there will always be residual amounts entrapped in the aquifer.

4. Limitations and Complexity of Groundwater Cleanup

Cleaning up contaminated groundwater always requires a lengthy operation period, such as up to several decades, and involves a substantial expense. Even with a sizable remediation cost and effort, failure to meet the cleanup goal is not uncommon. The U.S. National Research Council listed five key technical reasons for the difficulty in groundwater cleanup[2]:

4.1. *Physical Heterogeneity*

The subsurface environment is highly heterogeneous and often stratified with layers of materials with vastly different properties, such as clay, sand, and gravel over rock, and even within a layer the composition may vary over distances as small as a few centimeters. Fluids can move only through the permeable layers such as sands and gravels or through fractures in rocks and clays. Because these permeable flow channels are distributed nonuniformly, subsurface contaminant migration pathways are often extremely difficult to predict.

4.2. *Presence of NAPLs*

The entrapped small immobile NAPL globules in soil pores cannot be removed by flushing but can dissolve in and contaminate the passing groundwater. Removing DNAPLs is further complicated by their tendency to migrate deep into underground, where they are difficult to detect. Because of their high interfacial tension and their low aqueous solubility, NAPLs are extremely difficult to remove from an aquifer by applying conventional remediation techniques such as PAT. They may remain in pools, dissolve slowly, and act as a continuous source for contaminant plumes over a very long time.

4.3. *Migration of Contaminants to Inaccessible Regions*

Contaminants may diffuse to regions inaccessible by the flowing groundwater. Such regions may be microscopic or macroscopic. Contaminants present in these regions can serve as long-term sources of pollution, as they slowly diffuse back into the cleaner groundwater.

4.4. *Sorption of Contaminants to Subsurface Materials*

Most contaminants have a tendency to adhere to various solid materials in the subsurface. These contaminants can remain in place for long periods and then be desorbed or dissolved when the contaminant concentration in the groundwater decreases.

4.5. *Difficulties in Characterizing the Subsurface*

The subsurface cannot be accessed or examined in its entirety, but is observed through a finite number of drilled holes, localized excavations, and monitoring installations. Because of the highly heterogeneous nature of subsurface properties and the spatial variability of contaminant concentrations, observations from limited sampling points or locations will lead to incorrect interpolation and extrapolation, and inevitably an incomplete knowledge of subsurface characteristics.

The above inherent complexities pose major obstacles to groundwater cleanup, regardless of the remediation technology chosen. A good characterization of the contaminated aquifer, source zone, and the plume distribution may help to reduce the level of uncertainty in groundwater cleanup.

5. Site Investigation for Groundwater Cleanup

Site characterization is crucial to the success of the groundwater cleanup effort. According to the U.S. National Research Council (NRC), the poor characterization of the extent of contamination and the locations of contaminant sources resulted in the failures of several site cleanup systems.[2] Designers must consider not only the type of contaminant and the area to be remediated, but also the local hydrogeologic conditions, the available areas to construct a treatment system, location of utilities below the ground surface, the realistic cleanup goal to be achieved, and the impact on other treatment systems located nearby. Typically, the site characterization studies must be able to provide the following information:

- The contamination source, the extent of groundwater contamination, plume area, the contaminants of concern and their prevailing phases, migration pathways, and concentrations;
- The site geologic and hydrogeologic settings important to the design of the remediation system;
- The site constraints that may affect design and operation of the remediation system; and
- The recommendation of the most suitable remediation technology.

A good site characterization is labor- and capital-intensive, but worthwhile as the information collected assists in designing the most cost-effective remedial system and, therefore, reduces the remedial cost. Figure 6 illustrates the relationship between sampling intensity and remediation cost. Therefore, an adequate

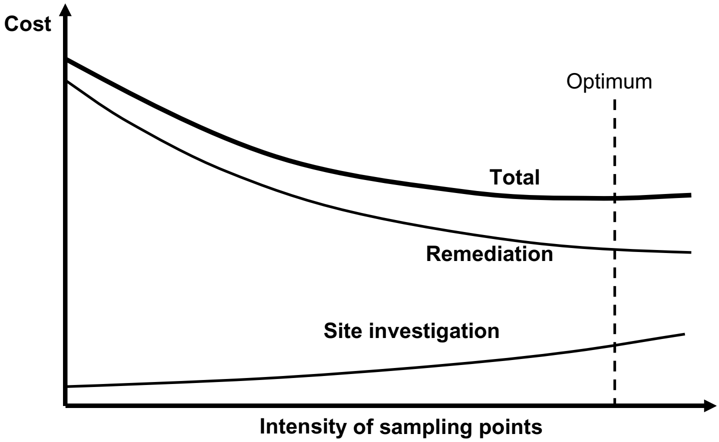

Figure 6. Relationship Between Sampling Intensity and Cost of Remediation.

number of drillings and samplings must be carried out at suspected areas. Samples collected include soils, vadose zone gas and water, and groundwater. A good field sampling plan and quality assurance project plan (QAPP) should be used. A monitoring program is also an integral part of a site investigation. Monitoring wells that allow sampling at various depths should be installed. A monitoring plan should be included in a QAPP, and it should emphasize the monitoring network design, monitoring well construction, data collection methods, and monitoring frequency. As much as possible, site characterization should provide the correct delineation of NAPL location or other source zones, and contaminant plume distribution and movement.

6. Groundwater Cleanup Strategy

Defining a cleanup strategy at the beginning of any cleanup project is important for ensuring effective remediation of a contaminated site without over-spending. The overall goal of groundwater cleanup essentially is to protect current and future groundwater resources and to avoid negative impact to ecosystems. A cleanup strategy may include a range of objectives such as limiting the migration of contaminant plume off-site, isolating and containing the source area, and finally treating the contaminated groundwater down to acceptable levels.

Groundwater cleanup planning should begin with the identification of a target treatment area. The location of a target treatment area depends on the contamination type, distribution, and the proximity to receptors. Source removal is a key component of most groundwater remediation projects. Many practitioners advocate targeting the source zone for the remediation of petroleum-contaminated aquifers. The presumption is that if the source zone can be located and removed or contained, then the remaining dissolved plume will dissipate through a natural attenuation processes. However, for remediation of contamination caused by DNAPLs, remediation outside the source area is warranted to prevent their spreading, as these compounds are biorefractory and their natural attenuation processes are very slow. Furthermore, the source zones of DNAPLs may be relatively large and difficult to locate due to the sinking and spreading of DNAPLs in the aquifer.

The U.S. EPA's groundwater strategy takes the following approach[8]:

- Focus resources at facilities that warrant attention in the near term.
- Control short-term threats.
- Prioritize actions within facilities to address the greatest risks first.
- Make progress toward the ultimate goal of returning contaminated groundwater to its maximum beneficial use.

According to the U.S. Environmental Protection Agency (EPA), this strategy benefits the public because it promotes early actions and continued progress toward the overall groundwater protection and cleanup goals. Regulators benefit because it helps them focus their resources on defining, tracking, and, if necessary, enforcing measurable milestones. Facilities benefit because the strategy helps them plan for investigation and cleanup actions.

7. Setting a Cleanup Level/Goal

There are many approaches that can be used to establish a cleanup goal. These include:

- drinking water standards,
- groundwater use designation's cleanup level or other government regulations,
- background level of the uncontaminated groundwater,
- concentrations of contaminants in adjacent surface waters,
- analytical detection limits,
- risk-based cleanup level,
- technology-based cleanup level, and
- cost-based cleanup level.

Fundamentally, the goal of groundwater remediation must be to protect the public health and environment. The question of "how clean is clean?" is one that has been debated by scientists, engineers, and legislators for decades. In the past, drinking water standards were often used in developing groundwater cleanup levels. However, cleanup goals based on drinking water standards could not be met with available technologies in many cases, or required tremendous amounts of money and time.[2] Therefore, the cleanup goal must be set to reflect what can be feasibly achieved with current constraints without posing a risk to public health and ecology. The last decade has seen a movement from a "clean, no risks" objective toward the protection of public health and the environment through risk reduction. Nowadays, it appears that a risk-based approach is gaining popularity, especially in establishing alternate concentration limits. This approach can be also used to develop a groundwater cleanup level for a constituent whose cleanup standard is not available. The fundamentals involved in determining risk-based cleanup level are assessment of all possible exposures to the contaminants, modes of exposure, magnitude of exposure, and the toxicity of the contaminant to the receptors. Several advantages and weaknesses of risk-based methods are given below.

7.1. *Advantages*

- A systematic approach is carried out.
- It is more versatile and applicable for all scenarios.
- It promotes data collection.
- It promotes development and use of scientifically accepted fate and transport models.
- It prioritizes sites for corrective action.

7.2. *Weaknesses*

- It does not necessitate complete contaminant removal from the contaminated media, and thus contaminants are more likely to remain in place after cleanup.
- There are several uncertainties such as contaminant sources, migration pathways, receptors, exposure levels, toxicity of compounds, and dose-response models.
- It does not drive the development of innovative treatments because current technologies are able to meet the risk-based cleanup levels.

8. Technologies for Groundwater Cleanup

Remediation of contaminated groundwater is often carried out on site. In the remediation of contaminated aquifer, often more than one cleanup method will work. Some of these technologies involve actual *in-situ* treatment — treat the groundwater in place without extraction followed by above-ground treatment. Others require the extraction of groundwater or contaminant vapor released from groundwater for above-ground treatment. The selection criteria include:

- Cost-effectiveness of the treatment technology in treating the contaminants of concern;
- Applicability to the target treatment area — contaminant source (such as NAPL) or contaminant plume (dissolved phase);
- Applicability to site geology and hydrogeology;
- Treatment duration to meet cleanup standards;
- Technology maturity, availability of expertise, and past experience;
- Simplicity in operation and maintenance; and
- Regulatory acceptance of the technology.

There are dozens of cleanup technologies that have been implemented in full-scale groundwater remediation and cleanup. Among the most common ones are:

- Containment method,
- PAT,
- Surfactant/cosolvent flushing,
- Air sparging (AS),
- Permeable reactive barrier (PRB),
- *In-situ* groundwater bioremediation,
- Monitored natural attenuation (MNA),
- Phytoremediation,
- Hydraulic and pneumatic fracturings,
- *In-situ* thermal treatment, and
- *In-situ* chemical oxidation (ISCO).

Besides these technologies, there are several emerging innovative groundwater cleanup technologies that show promising performance for full-scale application. The following sections describe the basic principles, advantages, and disadvantages of each of the technologies listed above.

8.1. *Containment Method*

8.1.1. *Description*

Containment is a technology used to isolate the contaminated zone, particularly the sources, from surrounding groundwater for preventing pollution diffusion and dispersion caused by flowing water, gasification, volatilization, or evaporation using engineered barriers. The sources of contaminants that are usually contained include waste landfills and NAPL pools. The containment method is a favorable solution over other alternatives where the waste mass is too large for treatment and where soluble and mobile constituents pose an imminent threat to a source of drinking water. In addition, the presence of complex contaminant distribution in subsurface may warrant a containment system to confine the contaminated area. The contaminant method has been also implemented to divert contaminated groundwater from the drinking water intake, divert uncontaminated groundwater flow, and provide a barrier for the groundwater treatment system. The successful implementation and effective containment system for these purposes require correct information on the source zones, groundwater flow patterns, and size, shape, and directions of movement of contaminant plumes. A containment system may also be used in combination with other remediation systems. In many cases, barriers can be installed very quickly, thus providing additional time before remedial action is required.

8.1.2. *Types of Containment Systems*

Engineered barriers can be vertical or horizontal. They can be as simple as a single vertical containment wall surrounding a contaminated area or as complex as a vertical wall system combined with a horizontal barrier or capping at an appropriate depth to provide complete and below-ground containment. Surface containment or capping, normally consisting of an impervious cover system, is designed and constructed to prevent water infiltration. It is an effective barrier for controlling leaching of waste or vertical influx of contaminant from a pollution source that can subsequently contaminate the underlying aquifer. The impervious materials commonly used for capping include clay, geomembrane, or composite geosynthetic clay liners. Vertical containment or a vertical cutoff wall, which controls the horizontal migration of pollutants, is perhaps the most common and important element in a containment system. It is constructed to control spatial distribution of a contaminant plume by creating a cutoff barrier in the ground using a slurry wall, grout curtain, sheet pile cutoff wall, or a composite wall with an High Density Polyethylene (HDPE) membrane. In case of deficiency or absence of an impermeable substratum at the base, a deep horizontal containment is built to provide a plug at the base of the vertical containment system using various sealant injection and jet grouting techniques.

The effectiveness of a containment system depends on several factors, such as:

- Hydrologic characteristics of the site,
- Climatic characteristics of the site,
- Geologic characteristics of the site,
- Characteristics of the contaminant plume or source being contained,
- Accuracy in understanding the boundaries of contaminant plumes and sources,
- Appropriateness of design, e.g., configuration and integrity of wall or caps,
- Quality assurance practices during construction,

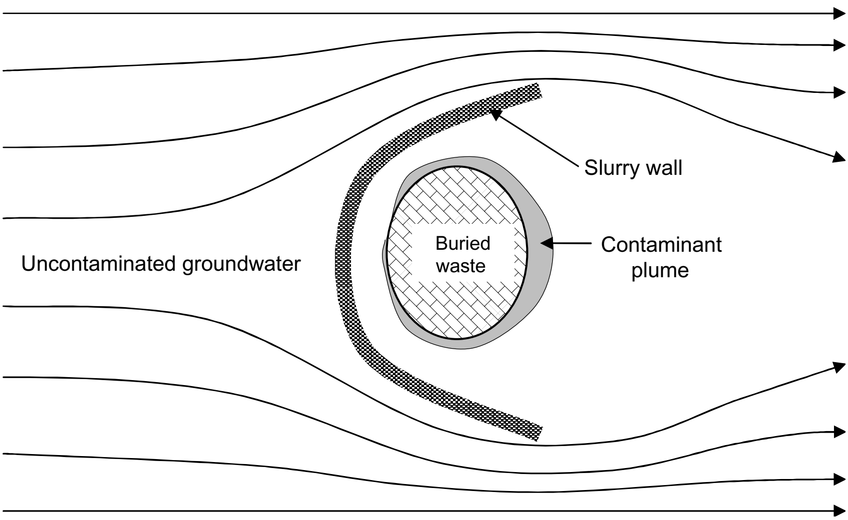

Figure 7. Plan View of an Arc-Shaped Vertical Slurry Wall Barrier for Diverting Uncontaminated Groundwater Around Buried Waste.[9]

- Material properties such as strength, compatibility, impermeability, and durability, and
- Maintenance frequency and effectiveness.

There are several possible layouts and configurations of the containment systems, with the main variance in the vertical barriers. The vertical barriers can be of any alignment, so long as it is strategically placed to effectively serve its purposes. An example of a possible layout of containment system that diverts uncontaminated groundwater flow from coming into contact with buried waste is shown in Fig. 7. Due to the cutoff from groundwater, the contained waste only emits contaminants through diffusion. Therefore, the contaminant plume created is rather small.

8.1.3. *Design Requirements*

To design and construct an effective containment system, the following factors must be assessed: maximum allowable permeability, constructability, anticipated hydraulic gradients, required wall strength, boundaries of contamination, compatibility of wastes and contaminants in contact with wall materials, characteristics (i.e., depth, permeability, and continuity) of substrate into which the wall is to be keyed, characteristics of backfill material, and site terrain and physical layout.

If a slurry wall is used, the limitation in its constructability, especially with respect to depth, has to be accounted for. Slurry walls are typically placed at depths up to 30 m and are generally 0.6–1.2 m in thickness. The most effective slurry wall system for pollution control is to base (or key) the slurry wall 0.3–0.6 m into a low permeability layer such as clay or bedrock. This "keying-in" provides an effective seal at the bottom with minimum leakage potential. An alternate configuration for slurry wall installation is a "hanging" wall in which the wall projects into the groundwater table to block the movement of lower density or floating contaminants such as LNAPLs and gases.

Most slurry walls are constructed of soil-based backfills, which typically consist of soil, bentonite, and water mixture. Walls of this composition provide a barrier with low permeability and chemical resistance at low cost. Soil-based backfills typically have a permeability in the order of 10^{-7} cm/s. Other wall compositions, such as cement/bentonite, pozzolan/bentonite, attapulgite, organically modified bentonite, or slurry/geomembrane composite, may be used if greater structural strength is required or if chemical incompatibility between bentonite and site contaminants exists. During the past decades, high-strength and self-hardening slurry walls with extremely low permeability have been increasingly used in Europe and North America. A cement–bentonite mixture is the most commonly used self-hardening slurry. Self-hardened slurry walls typically have a permeability of 10^{-6} to 10^{-9} cm/s, depending on the mixtures.

Questions about the type of mixture and the optimum ratio of the components in the mixture often arise in the design of a slurry wall. These can be best determined in the lab prior to construction. Several factors should be considered when designing the slurry mix, including the availability of the materials and their cost, constructability, permeability of the finished wall, density, and possible influence of *in-situ* materials such as soil organic matter and contaminants on the resulted wall's performance. Based on an initial technical screening, bench-scale testing is often performed to confirm the optimum mix design that will meet the required performances. A field trial mix will then be carried out to ensure that design requirements can be met before installing the walls.

8.1.4. *Limitations*

Although the design of a containment system is considered relatively easier compared to other remedial systems, several factors may limit its applicability. These include:

- Most containment systems involve a large amount of heavy construction and site clearance.
- Generally, vertical cutoff walls cannot be installed at sites with underground obstacles.
- Vertical slurry walls are limited to a maximum depth of 50 m.
- Contaminants are not removed or treated; the technology contains only contaminants within a specific area, and, therefore, further remediation in the future may be necessary if leakage is detected.

- There is the potential for the slurry walls to degrade or deteriorate over time due to chemical attack.
- The long-term performance of the slurry wall is not proven.
- Regulatory concerns and the types and concentrations of the contaminants at the site over time drive the duration of the monitoring, which may last for several years.
- The groundwater flow pattern downstream as well as immediately upstream may be altered permanently.

8.2. *Pump-and-Treat*

8.2.1. *Description*

Pump-and-Treat (PAT) is a groundwater treatment process that pumps out groundwater from the contaminated aquifer through extraction wells to the surface where the contaminants are removed or destroyed. Thus, it is sometimes considered as an *ex-situ* treatment. The treated water is then discharged into surface water bodies such as rivers, ponds, or lagoons or recycled back into injection wells to replenish the groundwater reservoir. Figure 8 provides a schematic illustration of a PAT system in groundwater remediation. PAT was the most common form of groundwater remediation until the 1990s.

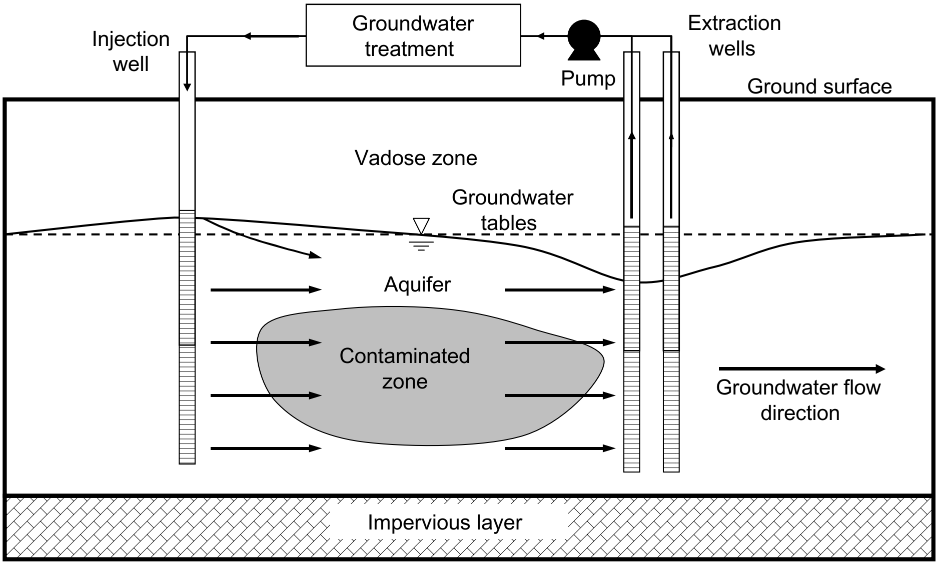

Figure 8. Schematic Illustration of PAT System with Reinjection of Treated Groundwater.

8.2.2. *Design of Pumping System*

The basic operating principle of a PAT system is to pump extraction wells at rates that cause all water in a contaminant plume to enter the well. Placement of the pumping wells and the pumping rates depend on the site characteristics and contaminant properties. The design elements of pumping patterns (systems) include the locations of extraction and injection wells, well depths, and pumping rates. Designing an appropriate system requires a good understanding of site hydrogeologic conditions, contaminant source, behavior, dissolved concentration, and areal extent, moisture precipitation, and groundwater recharge. The tasks that need to be accomplished before designing the pumping system are described in the following sections.

8.2.2.1. Aquifer characterization

The properties to be investigated include seasonal groundwater table, groundwater flow direction, hydraulic conductivity, transmissivity, and hydraulic gradient. A good understanding of hydraulic properties is essential for determining the suitable locations of pumping wells, pumping rates, and the number of wells required.

8.2.2.2. Geological characterization

The data that should be collected include soil types, moisture content, void ratios, bulk density, color, and the presence of layering or macrofabrics, which give information on the heterogeneity of the site geology.

8.2.2.3. Contaminant characterization

The essential information includes contaminant sources, types and concentrations of contaminants, contaminant solubility, volatilization, mobility and transport pathways, contaminant–aquifer interaction parameters, and likelihood of contaminant abiotic and biotic transformations.

8.2.2.4. Extent of contamination

Both the vertical and the horizontal extents of the contamination need to be known for designing the injection and extraction systems and performing capture zone calculations.

8.2.2.5. Groundwater analysis

Inorganic or mineral ions (calcium, magnesium, carbonates, and others) should be detected, as they can cause water hardness and mineral deposition.

There is increasing popularity in the application of numerous mathematical and groundwater transport models to provide insight into flow patterns generated by alternative PAT approaches and the associated travel times to extraction wells. These computation models are powerful tools for estimating plume movement and response to various pumping patterns. The various purposes of performing groundwater flow modeling for PAT remediation include the following:

- Predict pathlines and concentrations of contaminants at receptor points.
- Guide the placement of pumping wells and injection wells and monitoring wells.
- Guide the selection of monitoring points and frequency.
- Evaluate expected remedial performance under a variety of alternative designs so that the cleanup efficacy and time can be predicted and optimized.
- Evaluate the effect of source-control actions on remediation.

The decision to use modeling is made based on the objectives of the modeling, the ease with which the subsurface can be conceptualized mathematically, and the availability of realistic data. For relatively simple hydrogeologic settings such as homogeneous isotropic aquifers, analytical equations solved manually or graphically, or computer codes based on analytical solutions, may be adequate. For more complex sites, numerical computer models may be required. Several simple computer software, such as WHPA model[10] and Capture Zone Analytic Element Model (CZAEM),[11,12] both are developed by U.S. EPA, allow capture zone and groundwater pathline analysis. The numerical MODFLOW and MODPATH models developed by the U.S. Geological Survey are commonly used to model more complex hydrogeologic settings. Models such as the MT3D[13] and RT3D,[14] the complementary software to MODFLOW, can be used to simulate flow patterns and concentration changes resulting from the operation of a PAT system.

Optimization programming methods have been used increasingly in the past to improve PAT system design.[15] The advantage of using an optimization approach is that it allows selection of the best design based on the prerequisite objective and a set of constraints, such as minimizing the sum of pumping rates from a number of wells. The optimal solution method can be also coupled with artificial neural networks to facilitate searching of the optimal design, as demonstrated by Rogers *et al.*[16]

Although there have been a wide range of computation models developed that can perform very complex computation tasks, caution should be exercised when applying such models. There can be great uncertainty whenever subsurface transport is modeled, particularly when the results of the model are based on simplified conceptualization and estimated parameters. The predicted performance of PAT can be only as reliable as the quality of the input data and reliability of modeling assumptions.

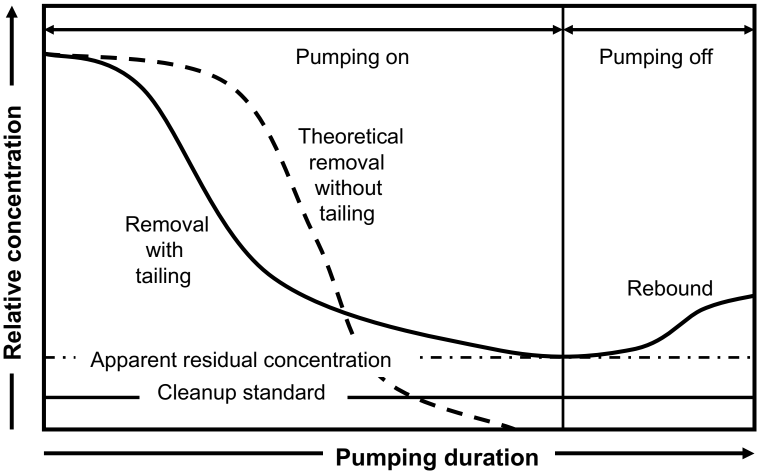

Figure 9. Concentration versus Pumping Duration Showing Tailing and Rebound Effects.[17,18]

8.2.3. *Tailing and Rebound*

The phenomena of tailing and rebound are commonly observed at PAT sites. *Tailing* refers to the progressively slower rate of decline in dissolved contaminant concentration with continued operation of a PAT system, while *rebound* is referred to increase in contaminant concentration that can occur after pumping has been discontinued. Both phenomena are depicted in Fig. 9. Tailing and rebound effects are problematic, as they can result in longer treatment time and residual concentrations (after long period of pumping and after pumping off) in excess of the cleanup standard.

The occurrence of rebound is attributed to the complexities of hydrogeology and contaminants. Complexities associated with hydrogeology include stratification, heterogeneities, the presence of fractured structures (such as fractured clays and rocks), and clay lens in aquifers. When such features are present, different zones of the contaminated aquifer will be flushed at different rates. Zones with high permeability will be cleaned up before zones with lower permeability. The latter may not be cleaned up at all due to flow by-pass. The regrowth of a contaminant plume may occur after pumping off when contaminants in zones of low permeability diffuse into the flowing groundwater.

The complexities associated with contaminants are irregular spatial distribution of contaminant plumes and sources, slow kinetics of contaminant dissolution, volatilization, degradation, and adsorption/desorption, and the presence of LNAPL and DNAPL. PAT can remove only the dissolved contaminants. When a portion of the contaminants mass remains sorbed to solid media or a portion of NAPL remains trapped in the dead zones, the PAT may not meet the cleanup standards as the remaining contaminants will continue to slowly dissolve into the groundwater after pumping is discontinued. It is commonly understood that as the complexity of the hydrogeologic properties and contaminant chemistry increases, the likelihood that the PAT system will meet stringent cleanup goals decreases. Therefore, deciding when to turn the system off is a crucial design and operation component in a PAT system.

8.2.4. *Cleanup Time*

Remediation by PAT is a slow process and cleanup times are often very lengthy. The rate of cleanup depends on contaminant variables and hydrogeologic variables such as well locations, transmissivity, dispersivity, hydraulic gradient, and pumping rates. A good estimation of cleanup time for PAT is difficult, hampered by complexities of hydrogeologic and contaminant properties, inability to control flow patterns between injection and extraction wells, and complex configuration of extraction and injection wells system. However, with reasonable assumptions, an estimation can still be made. For example, cleanup time, expressed as the number of required pore volume of the treated zone, can be roughly estimated using a batch flush model proposed by Zheng *et al.*,[19] which specifies that:

$$\mathrm{PV} = -\mathrm{R} \ln \frac{\mathrm{C_{wt}}}{\mathrm{C_{wo}}} \tag{2}$$

and

$$\mathrm{R} = \left(1 + \frac{\rho}{n}\mathrm{K_d}\right), \tag{3}$$

where PV = the number of pore volumes, R = retardation factor, $\mathrm{C_{wo}}$ = initial contaminant concentration in the groundwater (before PAT) (mg/L), $\mathrm{C_{wt}}$ = required cleanup concentration (mg/L), ρ = aquifer bulk density (kg/m^3), n = effective porosity of aquifer, and $\mathrm{K_d}$ = distribution coefficient (L/g).

The batch flush model is a useful approach for estimating cleanup times in a simple aquifer system with chemicals for which interaction with the solid matrix can be represented by linear sorption. In general, the batch flush model will underestimate cleanup time because it does not account for certain processes that can result in prolonged treatment time, such as geologic heterogeneity, the presence of NAPL, and leachate from remaining contaminant sources.[2]

8.2.5. *Treatment System*

Once extraction wells have brought contaminated water to the surface, treatment is relatively straightforward, using conventional industrial wastewater treatment and water treatment technologies. Technologies for the treatment of contaminated groundwater fall into two main categories, which are biological treatment methods

Table 2. Applicability of Various Treatment Technologies to Contaminated Groundwater.[20]

Contaminants	Neutralization	Precipitation	Coprecipitation/Coagulation	UV/Ozone	Chemical Oxidation	Reduction	Distillation	Air Stripping	Steam Stripping	Activated Carbon	Evaporation	Gravity Separation	Flotation	Membrane Separation	Ion Exchange	Filtration	Biological	Electrochemical
Heavy metals	X	●	●	X	X	○	X	X	X	○	●	●	X	●	●	●	X	●
Chromium(VI)	X	●	X	X	X	●	X	X	X	○	●	X	X	○	●	X	X	●
Arsenic	X	○	●	○	○	X	X	X	X	○	X	○	X	●	●	●	X	X
Mercury	X	●	●	X	X	●	X	X	X	●	X	○	X	○	●	●	X	X
Cyanide	X	X	X	●	●	X	X	X	X	X	●	X	X	●	●	X	○	○
Corrosives	●	●	X	X	X	X	○	X	X	X	X	X	X	X	X	X	X	X
VOCs	X	X	X	○	●	X	●	●	●	●	X	X	X	○	○	X	○	X
Ketones	X	X	X	○	●	X	●	●	●	X	X	X	X	X	X	X	●	X
SVOCs	X	○	○	●	●	X	●	X	●	●	○	○	○	●	●	X	●	X
Pesticides	X	○	○	●	●	X	●	X	○	●	○	○	○	●	●	●	○	X
PCBs	X	●	●	●	●	X	●	X	X	●	●	●	●	●	●	●	○	X
Dioxins	X	●	●	●		X	●	X	X	●	●	●	●	●	●	●	○	X
Oil and grease/ floating products	X	●	●	X	X	X	●	X	X	X	●	●	●	●	●	○	○	X

Notes: ● Applicable, ○ Potentially Applicable, and X Not Applicable.

and physical/chemical treatment methods. Biological treatment methods may be applicable for the treatment of groundwater contaminated by organic compounds if their concentrations are not above levels which could be toxic to microorganisms. Physical or chemical methods, or a combination of both, commonly involve adsorption, separation, precipitation, and oxidation processes. Table 2 summarizes the applicability of various treatment technologies to groundwater contaminated by any of the major categories of inorganic and organic contaminants.

8.2.6. *Advantages and Limitations*

A key advantage of the PAT method is that it is borne out by much professional experience with hydrogeology. The pumping technologies themselves are also well proven and efficient. However, most PAT remediation projects have a high degree of uncertainty in term of cleanup up time and ability to meet cleanup goal. Experience gathered over the last decade indicates that returning the groundwater to drinking water standards may not be possible at many sites with PAT systems within a reasonable time frame.[2] A conventional PAT may take decades or even centuries to clean up DNAPL contaminated groundwater. Because cost is largely based on the time required for cleanup, PAT is often not the most favorable choice for remediation.

8.3. *Surfactant/Cosolvent Flushing*

8.3.1. *Description*

The surfactant/cosolvent flushing, also known as surfactant/cosolvent-enhanced aquifer remediation technology, is an emerging technology for groundwater remediation that aggressively attacks NAPL source areas. This technique involves injection of cosolvent mixtures, surfactant mixtures, or surfactant/cosurfactant/cosolvent mixtures to the upgradient groundwater system at injection wells to mobilize and/or solubilize NAPL. Groundwater laden with the surfactant/cosolvent and the solubilized NAPL is subsequently extracted downgradient through strategically placed extraction wells, as depicted in Fig. 10.

Effective application of this technology depends on the ability to control *in-situ* flows. Physical barriers may be installed to prevent uncontrolled migration of surfactant/cosolvent and contaminants. Complex and heterogeneous geology and the presence of low permeability zones increase the difficulty of applying this

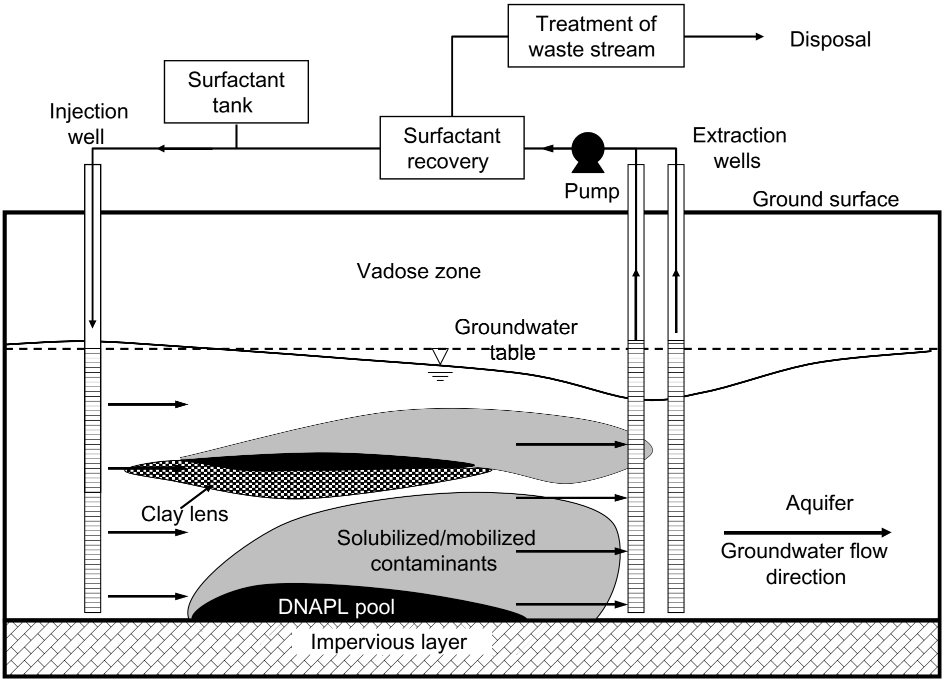

Figure 10. Schematic Illustration of Surfactant-Enhanced Removal of DNAPLs.

technology. Thus, the technology is best applied to moderate to high permeability geologic materials. It is used most often when the groundwater is contaminated by DNAPLs such as trichloroethylene (TCE) and tetrachloroethylene (PCE). Aboveground treatment processes are typically carried out to treat the produced fluid and recycle the surfactant if necessary.

8.3.2. *How Cosolvents and Surfactants Work*

8.3.2.1. Cosolvents

Cosolvents are substances that, when mixed with water, enhance the solubility of hydrophobic organic compounds that are otherwise insoluble in water. Examples are alcohols (such as methanol and ethanol) and ketones (such as acetone). Cosolvents can promote contaminant removal in two ways. The first is by increasing the apparent solubility of the contaminant in water, which improves the mass removal per pore volume. The second is by reducing interfacial tension between the water and the contaminant, which may result in direct mobilization of NAPL. The solubility enhancement by cosolvency is determined by fraction of the cosolvent in the cosolvent mixture and by the solubility of the solute in the cosolvent.

8.3.2.2. Surfactant

Surfactant monomers have both polar and nonpolar groups that exhibit hydrophilic and hydrophobic properties, respectively. Polar groups may be ionic (anionic or cationic surfactants) or nonionic. Because of the dual nature of surfactant molecules, surfactants tend to accumulate at the surface or interface. As a result, they can achieve a significant reduction in the interfacial tension between the NAPL and water phases, so that NAPL solubilization or mobilization is greatly accelerated.

At a certain characteristic concentration, i.e., the critical micelle concentration (CMC), aggregations of surfactant monomers occur and form colloidal clusters of surfactants called micelles, with hydrophobic nonpolar organic tails in the center and hydrophilic polar groups on the outside. When added to oils or NAPLs, the micelle's exterior (polar groups) interfaces with the water molecules, while the hydrophobic groups are arranged pointing toward the interior of the micelle that contains the hydrophobic contaminant molecules. A schematic representation of this process is given in Fig. 11. This process can enhance the apparent solubility of NAPL by several orders of magnitude. The microemulsion formed has approximately the same density as water, which reduces the risk of uncontrolled spreading of the pollution to deeper, uncontaminated layers. With surfactants, NAPL removal can be accomplished in months, compared to years or decades with conventional PAT systems.

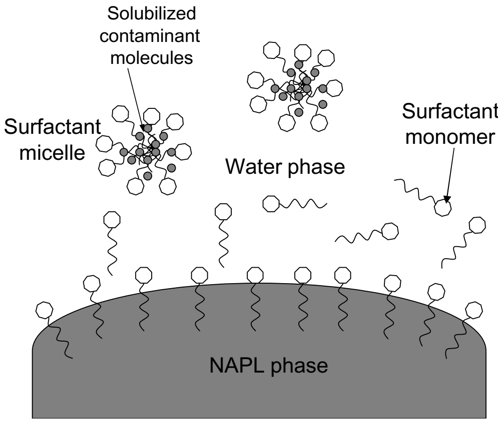

Figure 11. Surfactant Monomers and Micelles in Equilibrium with Contaminant Molecules and NAPL Phase.

There are in fact two main mechanisms that cause the removal of residual DNAPL globules by application of surfactants: (1) solubilization, which enhances the apparent solubility of the organic molecules in the DNAPL globules through the formation of micelles in groundwater (as discussed above), and (2) mobilization, which by achieving ultra-low interfacial tension (and thus lowering capillary forces acting on DNAPL globules), surfactants can mobilize bulk DNAPL in water. Both solubilization and mobilization can improve subsurface remediation in surfactant flushing. However, the mobilization of DNAPL in aquifers is not always desirable because of concerns of uncontrolled vertical movement of DNAPL.[21]

8.3.3. *Factors Affecting Performance of Surfactant/Cosolvent*

Parameters such as surfactant/cosolvent type, pore-water velocity, particle size, and surfactant/cosolvent concentration can considerably influence the solubilization of DNAPLs. The type of surfactant/cosolvent influences solubilization characteristics such as equilibrium solubility and mass transfer rate between the nonaqueous and the aqueous phases. Water velocity (which can be controlled by pumping rate) determines the contact time, and thus affects the amount of nonequilibrium interfacial mass transfer between mobile and immobile water regions. Particle size determines the pore velocity of the water, the size of DNAPL globules, and interfacial area. Surfactant/cosolvent concentration influences the equilibrium solubility of contaminants in groundwater. Sometimes, the mixtures of surfactants and cosolvents are used or salts are added to surfactant solution to improve the uptake of NAPL or the hydraulic properties, such as the viscosity of the surfactant solution.

8.3.4. *Surfactant Selection*

There are more than 12,000 types of surfactants.[9] They can be categorized into four major categories based on the chemical structure of their hydrophilic groups: anionics, nonionics, cationics, and amphoterics. When choosing a surfactant, several factors must be considered. These include surfactant effectiveness, biodegradability and toxicity, surfactant–soil interaction, interfacial tension between the water and NAPL, the solubility of the contaminant in the groundwater, hydraulic parameters, and the ability to cost-effectively manage the produced fluids. Some surfactants are readily adsorbed by mineral surfaces. Cationic surfactants are often adsorbed more than anionic and nonionic surfactants. The adsorption of surfactants by aquifer materials can lower the concentration of free surfactant for NAPL dissolution. In addition, adsorption will cause retardation in surfactant velocity. The adsorbed surfactant may eventually be remobilized at the end of surfactant flushing that may then pose potential environmental hazards. On the one hand, a high biodegradability of surfactant is undesirable, as it will decrease the available surfactant for NAPL removal during surfactant flushing period and may also cause bioplugging of an aquifer. On the other hand, it is desirable to have the remaining surfactant biodegraded in the aquifer at the end of surfactant flushing to eliminate them. The toxicity of surfactant selected for use in surfactant flushing must be considered in the surfactant screening process. The surfactants used should be nontoxic, food-grade, and with favorable biodegradability. In general, cationic surfactants are more toxic than anionic and nonionic surfactants.[22]

A typical screening study for a field application may involve evaluating more than 100 surfactants.[22] Previous experiences at sites with similar types of NAPL may help to identify the appropriate surfactants, but a surfactant screening study should be carried out for each site because groundwater and aquifer geochemistry can significantly influence surfactant performance. With available published literature of many case studies, recommendations from surfactant manufacturers, and site-specific studies, it is should not be difficult to determine the best possible surfactant to apply to different contaminated sites for remediation. Porter gives a comprehensive coverage of surfactants and provides practical guide to their selection.[23] A monograph of surfactant/cosolvent flushing in NAPL removal is also available.[24]

8.3.5. *Treatment and Surfactant/Cosolvent Recycling*

Surfactants and cosolvents are expensive, making the recycling of the flushing solution essential in a surfactant/cosolvent flushing system for maximum cost effectiveness. Most often, produced fluids are treated aboveground on site. Due to the

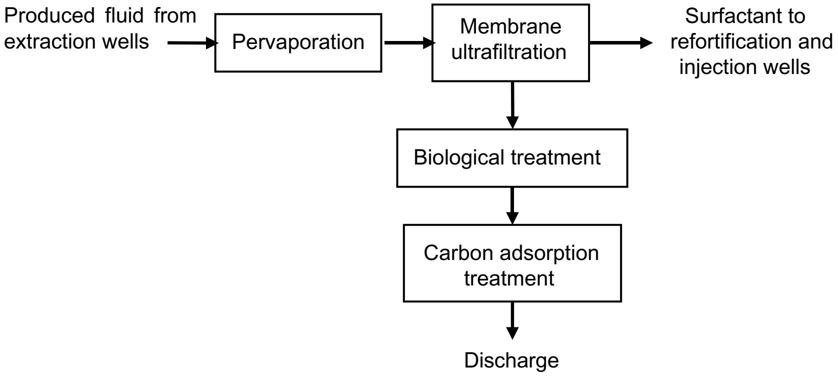

Figure 12. Treatment of Produced Fluid from Surfactant Flushing.

special characteristics of the surfactant/cosolvent system, the management of produced fluids is complicated. The produced fluids treatment trains may include pervaporation, gravity separation, dissolved-air flotation, chemical demulsification, acid treatment, salt addition, heat treatment, air stripping, steam stripping, membrane separation, biological treatment, solvent extraction, oxidation, and granular activated carbon (GAC) adsorption. Solvent extraction, membrane separation (particularly ultrafiltration), and biological treatment have been evaluated to the greatest extent and may be the most applicable treatment methods for semi-volatile contaminants.[22] For removing volatile contaminants, pervaporation and air stripping may be the most applicable technologies. Ultrafiltration may be needed to reconcentrate the surfactant if reuse is desired.[22] Finally, GAC may be used as a polishing step to remove the residual contaminants as well as surfactant/cosolvent after pretreatment with other technologies.

An example of produced fluid treatment train for surfactant flushing application is shown in Fig. 12. The recovered surfactant-DNAPL produced fluids are first treated using pervaporation to remove volatile DNAPL components from the water and surfactants. The pervaporation-treated waste stream is then passed to a membrane ultrafiltration unit to concentrate the surfactants. Once the surfactants have been separated from the groundwater, they are recycled back to the injection wells. The ultrafiltered water has to be further treated to remove contaminants using one of or a combination of several industrial wastewater treatment methods, such as biological treatment and GAC adsorption.

8.3.6. *Advantages and Disadvantages*

Currently, only a few alternatives are available to remediate DNAPL source areas. Surfactant/cosolvent flushing provides a solution to this challenge. This surfactant/cosolvent-based technology is cost competitive with conventional PAT systems and provides for a more rapid and effective site cleanup. Cosolvents that are also substrates for microbes may have the added advantage of promoting contaminant degradation by cometabolism if the cosolvents are applied at nontoxic levels. The main disadvantage of this technology is that if the contaminant plume resulting from NAPL dissolution is not captured effectively by the extraction well systems, the surfactant/cosolvent solution can worsen the extent of the contamination and increase the solubilized contaminants in groundwater. Certain surfactants and cosolvents are toxic contaminants, although in general they cause less pollution and decompose faster than NAPL. One of the main surfactant toxicity concerns is their impact on aquatic organisms[25]; they may be toxic to fish at the ppm levels. There is also concern that metabolites of alkylphenol ethoxylates (APEOs), the nonionic surfactants, can mimic natural hormones and that the levels present in the environment may be sufficient to disrupt endocrine function in wildlife and humans.[26] Another issue is the low biorefractory of certain synthetic chemical surfactants. Because of these concerns, the use of surfactant/cosolvent-enhanced flushing is not as common as conventional PAT remediation. Several recent studies have turned to using biosurfactants instead of synthetic chemical surfactants to solubilize contaminants in soil and groundwater.[27,28]

8.4. *Air Sparging*

8.4.1. *Description*

Air-sparging (AS) is an *in-situ* remedial technology that reduces the concentrations of volatile constituents adsorbed to soils and dissolved in groundwater. This technology involves the injection of air into the subsurface saturated zone, creating air and water interfaces and enhancing mass transfer of volatile constituents from a dissolved state to a vapor phase. An AS system can be used to restore aquifer water quality at the source, throughout the plume, or as a barrier to control spread of contaminant plume.

The effectiveness of an AS system depends primarily on the volatility of the contaminants to be removed, air permeability of the soil, and hydrogeologic features of the site. Volatility determines the rate at which dissolved a constituent can be transferred from the dissolved phase to the vapor phase. Permeability of the soil determines the rate at which air can be injected into the saturated zone and, therefore, the mass transfer rate of the constituent to the vapor phase. Hydrogeologic features include depth of groundwater, hydraulic conductivity, stratification, heterogeneity, and anisotropy. An AS system is best suited to sites with sandy soils and medium to shallow aquifer depths of less than 15 m below ground surface.[29] If the subsurface is relatively homogenous, the airflow distribution pattern will tend to form a

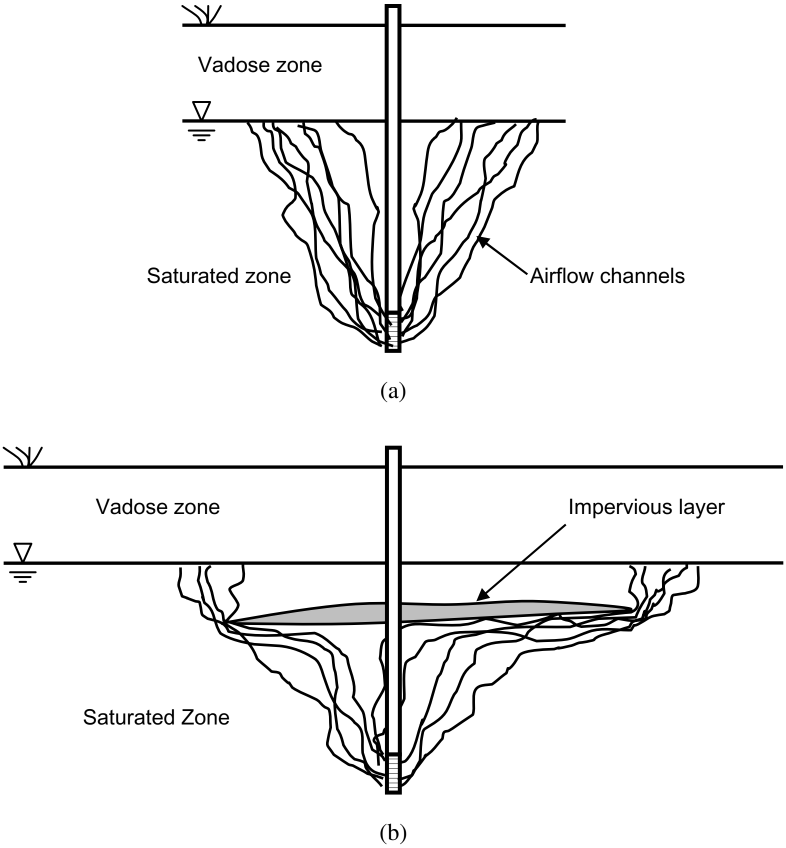

Figure 13. Effect of Subsurface Heterogeneity on Injected Air Distribution During AS: (a) Homogeneous Subsurface Geology and (b) Heterogeneous Subsurface Geology.

symmetrical and conical shape as shown in Fig. 13(a). Stratification, heterogeneity, and anisotropy will prevent uniform airflow through the medium, thus reducing the efficacy of AS. The presence of low permeability layers will make both the injected air and the stripped vapors traveling along the paths of least resistance (coarse-grained zones), and possibly a great lateral distance from the injection point (Fig. 13(b)). In general, sites having high clay or silt content in soils with hydraulic conductivities less than 1×10^{-3} cm/s are not recommended for this technology.

Bass *et al.*[30] reviewed case studies of 49 AS sites of chlorinated solvents and petroleum hydrocarbons contamination. They reported that groundwater contaminant concentration reductions ranged from 27% to almost 100%, with about half of the sites achieving greater than 90% reductions in contaminant concentrations. The varying performances could be due to different site hydrogeologic conditions, AS designs, construction techniques, and operational conditions.

8.4.2. *Contaminant Removal Mechanisms*

Volatile contaminants are removed by AS from groundwater primarily through air stripping. Lighter gasoline constituents such as benzene, toluene, ethylbenzene, and xylene (BTEX) and chlorinated hydrocarbons such as trichloroethylene (TCE), pentachloroethylene (PCE), dichloroethylene (DCE), and vinyl chloride (VC) are amenable to removal by AS because they readily transfer from the dissolved phase in the saturated zone to the gaseous phase in the unsaturated zone. These compounds have relatively high Henry's law constants and thus can be easily volatilized via AS. As a rule of thumb, compounds with a Henry's law constant above 1×10^{-5} atm $\cdot$ m^3/mol usually exhibit good strippability by AS. The stripped compounds are then removed via soil vapor extraction (SVE) and/or biodegradation in the unsaturated zone. Figure 14 illustrates a conceptual diagram of AS with SVE to extract vaporized contaminants.

Semi-VOCs (SVOCs) such as constituents in jet fuel, diesel fuel, and kerosene are more difficult to remove by air stripping. However, oxygen introduced by AS to the groundwater can stimulate aerobic biodegradation of these compounds. This is especially warranted if the initial dissolved oxygen (DO) level in the groundwater is less than 2 mg/L that is unfavorable for natural aerobic degradation. The enhanced aerobic biodegradation is the primary removal mechanism for SVOCs and the secondary removal mechanism for VOCs in an AS system.

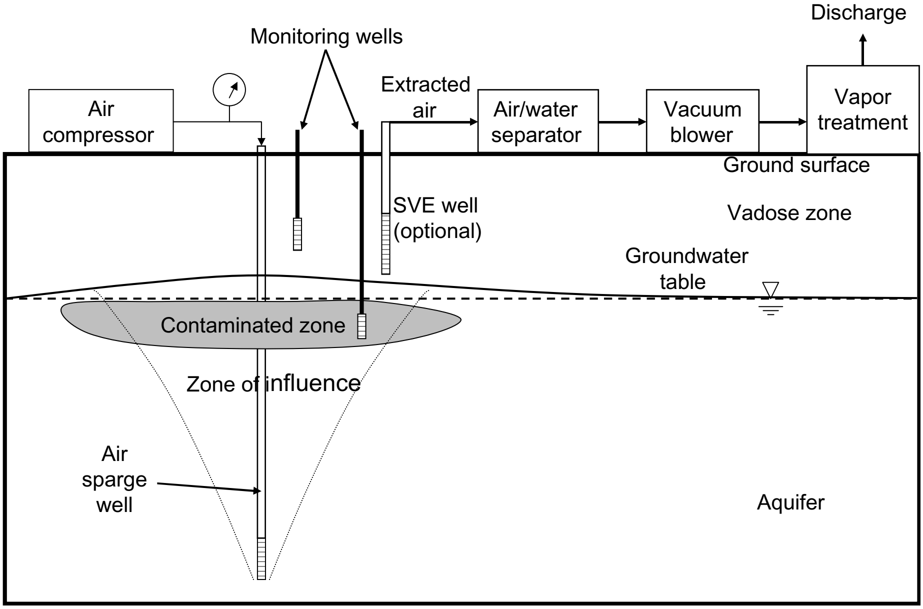

Figure 14. Process Diagram of an As System.

Because of the two aforementioned operative contaminant removal mechanisms, AS can be designed and implemented in the following ways:

- AS with SVE for removal of VOCs.
- AS without SVE but with enhanced biodegradation in vadose zone for removal of VOCs.
- AS without SVE but with enhanced biodegradation in saturated zone for removal of SVOCs.

8.4.3. *Determination of Target Treatment Area*

The target treatment area is the area where an AS system is installed to achieve the most effective treatment. It may encompass the source zone, the dissolved plume, localized areas with elevated concentrations within the plume, or the downgradient boundary of the dissolved plume.[29] For remediation of a petroleum-contaminated site in which the target compounds are predominantly VOCs and are biodegradable under aerobic conditions, it appears that the most economical AS installation may be source zone remediation. If natural attenuation is not sufficient to control plume migration before the plume contacts potential receptors, a sparge curtain or wall can be installed at the leading edge of the plume (Fig. 15). For remediation of chlorinated solvent contamination, targeting the localized areas with elevated concentrations within the dissolved plume and/or location of the leading edge of the plume is advocated. Treating the whole contaminant plume of DNAPLs may be economically prohibitive as the plume of DNAPLs such as TCE and PCE used to be very large.

8.4.4. *Pilot Studies*

To date, AS system design has been largely empirical, with significant variability in approaches and results.[32] Therefore, field pilot studies are necessary to provide

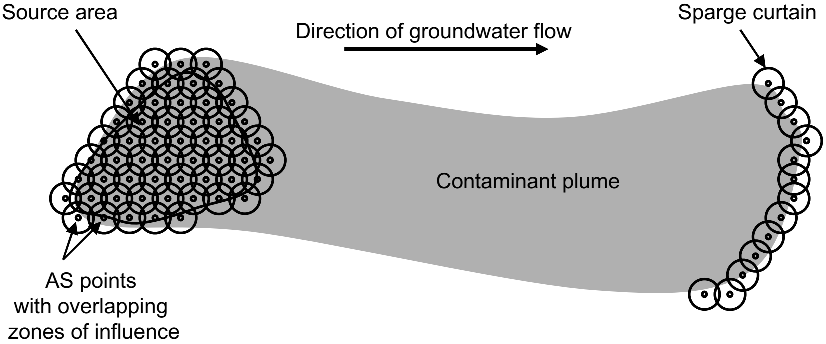

Figure 15. AS Point Locations in a Source Area and Sparge Curtain at the Leading Edge of the Contaminant Plume.[31]

data for the design and evaluating efficacy of an AS system. Pilot testing should be conducted in a portion of the target treatment area. The AS pilot test is designed to[29]: (a) assess feasibility; (b) characterize the air distribution pattern; (c) evaluate contaminant removal rates; (d) identify unexpected challenges; (e) identify any safety hazards to be addressed in the full-scale design; and (f) provide data to size the full-scale system. To accomplish these objectives, the following activities are typically carried out during pilot-scale testing:

- Baseline sampling prior to AS startup,
- Injection pressure and flow rate testing,
- Monitoring of vadose zone pressure changes,
- Monitoring of groundwater pressure response and elevations,
- Monitoring changes in contaminant concentration in groundwater,
- Soil vapor sampling and off-gas sampling (with SVE) for measurement of contaminant vapor concentrations in extracted vapors,
- Measurements of DO in groundwater,
- Tracer testing to indicate preferential flowpaths of injected air, and
- Direct observations of factors exist that make the system operation less acceptable (e.g., look for vapor migration to sensitive receptors).

8.4.5. *Design of AS System*

The essential goal in designing an AS system is to configure the injection wells and SVE wells and monitoring points in such a way to:

- Optimize the influence zone of each well and the clusters of wells, thereby maximizing the removal efficacy of the system.
- Provide optimum vapor extraction points to ensure minimal migration of the vapor plume.
- Provide optimum monitoring points to ensure no undetected migration of either the dissolved phase or vapor phase plumes.

8.4.5.1. Air injection system design

An air injection system design encompasses the determination of number of wells and configuration, well construction details, well screen depth, air flow rate, pressure, and injection mode (e.g., pulsed or continuous). Air injection wells are placed at location within the target treatment area. The placement and number of air sparge points required to address the target treatment area are determined primarily by the permeability and structure of the soil, as these affect the sparging pressure and distribution of air in the saturated zone. An AS system can use either vertical or horizontal sparge wells. Well orientation should be based on site-specific needs and conditions. The design radius of influence (ROI) for AS wells is the most important parameter to be considered in the design of the AS system. The ROI will help determine the number and spacing of the sparging wells needed. AS wells should be placed so that the overlap in their radii of influence completely cover the target treatment area (as shown in Fig. 15).

The injection well screen begins 1.5 m below the target treatment area and is typically 0.6 m long. Well screens installed greater than 1.5 m below the target treatment zone may result in air bypassing the contaminated area. On the other hand, injection wells installed too shallowly will likely result in air channels not contacting the lower portions of the treatment area. The sparging air flow rate and pressure and the appropriate injection mode are site-specific, which can be determined during the pilot study.

8.4.5.2. SVE system design

SVE systems are sometimes installed in conjunction with *in-situ* AS systems to prevent migration of contaminant vapors to enclosed underground spaces and to recover the vapors. The main components of an SVE system are extraction wells, vacuum blower, air/water separator, and vapor treatment system. The vapor extraction rate should typically be 2–3 times the air injection rate and should maintain an adequate vacuum in the nearby soil vapor monitoring points.[29] The minimum number of SVE wells required, N_{SVE}, can be estimated by the following equation:

$$N_{SVE} = \frac{A_t}{\pi(R_{SVE})^2}, \tag{4}$$

where A_t = target treatment area (m^2) and R_{SVE} = ROI for SVE well (m).

The SVE wells can be located adjacent to an AS well or in between a network of AS wells. An additional option for SVE well placement is to co-locate the extraction well and AS injection well in the same borehole. This strategy provides a conservative SVE well spacing, but will reduce installation costs. In shallow applications, large plume areas, and locations under buildings or pavements, the horizontal vapor extraction wells are very cost effective and efficient for controlling vapor migration. The extracted vapor can be treated on site by processes such as thermal oxidation, catalytic oxidation, and GAC adsorption.

8.4.5.3. Monitoring network

In a typical AS system, the monitoring network comprises groundwater monitoring and vadose zone monitoring for both performance monitoring and compliance monitoring. Parameters of interest in groundwater monitoring include contaminant concentration, biodegradation byproducts, and occasionally DO, pH, and redox potential. Soil vapor parameters of interest are contaminant concentration and soil gas pressure. The location of monitoring wells should span from upstream of the treatment area, through the treatment area, to downstream of the area. Monitoring points for soil vapor sampling must be located between the injection well and any buildings that may be at risk to ensure that the structures are not impacted by vapor-phase contaminants. Sampling of ambient air just above ground surface may be necessary to detect vapor gas emission.

8.4.6. *System Operation*

AS systems can be operated in either a continuous or pulsed mode, but pulsed mode is thought to achieve more extensive air distribution.[29] Pulsing the air injection can minimize the decrease in water permeability of sparged zone caused by air injection. Pulsing also enhances mixing of groundwater resulting from formation and collapse of air channels during each pulse cycle. Studies suggest that a pulsed operation could increase contaminant mass removal by 20–30%.[32,33] The total cost for air compressors is also lower in pulsed mode.

One important aspect in system operation is to estimate the contaminant removal rate. To date, there are still no reliable methods for estimating groundwater cleanup rates by AS. Therefore, SVE off-gas concentrations should be monitored regularly for VOC levels and volumetric flow rate to estimate the mass of VOCs removed by the SVE system. The total VOC mass removed ($\mathrm{m_{SVE}}$) in the SVE off-gas stream can be estimated using Eq. (5):

$$\mathrm{m_{SVE}} = \sum_{i=1}^{n} \mathrm{C_i Q_i} t_\mathrm{i}, \tag{5}$$

where $\mathrm{C_i}$ = average concentration of VOCs in SVE off-gas stream within the time interval t_i, (kg/m^3); $\mathrm{Q_i}$ = average volumetric flow rate in SVE off-gas stream within the time internal t_i (m^3/day); t_i = time intervals over which the concentration and flow rate data are taken to be representative (days); i = counting variable; and n = total number of time periods summed.

8.4.7. *Advantages and Disadvantages*

There are many advantages of AS system as listed below:

- Application of the technology is widely recognized by the regulatory community as an effective remedial technology for removing volatile contaminants from groundwater.
- Equipments are readily available and implementation is relatively easy.

- Cleanup times are relatively short, typically taking less than three years to achieve cleanup objectives.
- If SVE is not necessary, only minimal operational supervision is required once the system is installed and no waste streams are generated that require storage, treatment, and discharge.
- The cost of treatment is relatively cheap compared to PAT systems.

Factors that may limit the applicability and effectiveness of the process include:

- There can be uncontrolled movement of VOC-laden vapor that may escape from the SVE system or accumulate below underground structures and in underground conduits that will cause potential hazard.
- Soil heterogeneity and stratification may cause some zones to be relatively untreatable.
- High air pressure may fluidize soil and, therefore, jeopardize the stabilities of nearby buildings or facilities.
- Confined aquifers are not suitable for the application of this technology because the injected air would be trapped by the saturated confining layer.
- No reliable field and laboratory data or theories to support AS design calculations; cleanup rates are difficult to predict without a field trial.
- AS is not effective for removing LNAPL layers greater than one-foot thick or removing DNAPL.

8.5. *Permeable Reactive Barrier*

8.5.1. *Description*

Permeable Reactive Barriers (PRBs) are an emerging technology for the *in-situ* treatment of contaminated groundwater. Instead of trying to remove the contaminated water from the subsurface for above-ground treatment, a PRB is essentially a permeable reactive treatment wall consisting of reactive media that intercepts the contaminated plume and immobilizes the contaminant or transforms the contaminant to a nontoxic or biodegradable form. Once installed, it is considered a passive *in-situ* treatment method. The wall can consist of either a continuous trench or a gate with a series of vessels or cells. The extent of treatment depends on the nature of the contaminant, the selection of the reactive material, the physical design of the treatment system, and natural site conditions.

There are various ways a PRB system can be designed to attenuate contaminants in groundwater. Degradation barriers facilitate reactions that break down contaminants in the plume into harmless byproducts either by stimulated biodegradation or abiotic transformation. Precipitation barriers react with contaminants to

Table 3. Contaminants Treatable by Reactive Materials in PRBs.[34]

Contaminant Group	Species
Methanes	Tetrachloromethane Trichloromethane Dichloromethane
Ethanes	Hexachloroethane 1,1,1-Trichloroethane 1,1,2-Trichloroethane 1,1-Dichloroethane
Ethenes	Tetrachloroethene Trichloroethene *Cis*-1,2-dichloroethene *Trans*-1,2-dichloroethene 1,1-dichloroethene Vinyl chloride
Propanes	1,2,3-Trichloropropane 1,2-Dichloropropane
Aromatics	Benzene Toluene Ethylbenzene
Others (organics)	Hexachlorobutadiene 1,2-Dibromoethane Freon 113 *N*-nitrosodimethylamine
Trace metals	Chromium Nickel Lead Uranium Technetium Iron Manganese Selenium Copper Cobalt Cadmium Zinc
Anion Contaminants	Sulfate Nitrate Phosphate Arsenic

form insoluble products that remain in the barrier as groundwater continues to flow through. Sorption barriers clean up groundwater through adsorption of contaminants onto the reactive media. The various contaminants that can be treated with a PRB are given in Table 3.

8.5.2. *PRB Designs and Configurations*

PRBs can be installed as permanent, semi-permanent, or replaceable units across the flow path of a contaminant plume. The two most common designs are the continuous trench system and funnel-and-gate system, which are shown in Figs. 16 and 17, respectively. The continuous trench system is simply a trench that has been excavated and then backfilled with reactive materials, allowing the water to pass through the barrier under its natural gradient. In the funnel-and-gate PRB, the impermeable funnel walls (which typically consist of interlocking steel sheet-pile wall or a cement/bentonite slurry wall) direct the groundwater flow toward the reactive treatment gate that contains reactive media. The reactive media are often bordered by

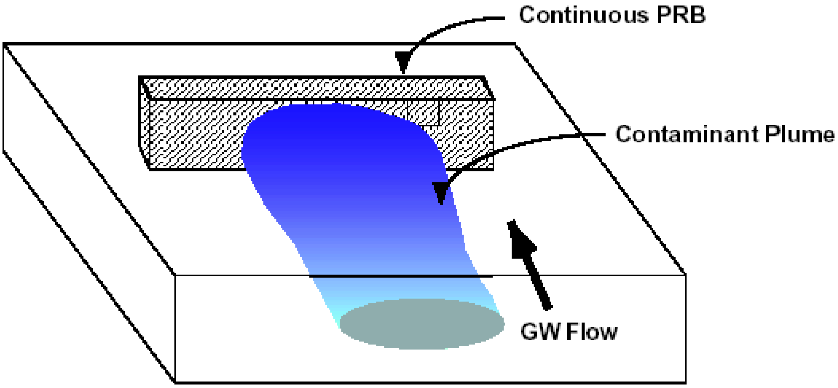

Figure 16. Schematic Illustration of Plume Capture by a Continuous Trenched Wall System.[34]

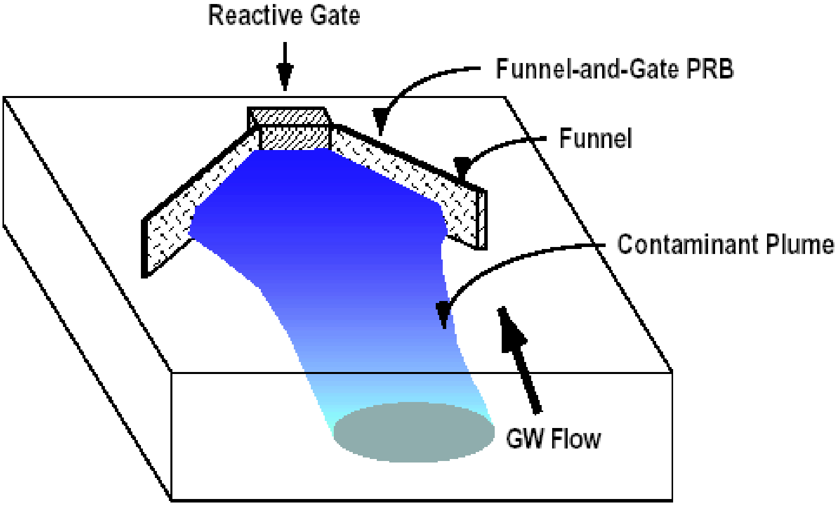

Figure 17. Schematic Illustration of Plume Capture by a Funnel-and-Gate System.[34]

sand or gravel, which increases the hydraulic conductivity of the gate and encourages groundwater and contaminants to disperse and flow uniformly through the gate.

To successfully clean up the plume, the PRB wall must be thick enough to allow long enough residence time for adequate reaction between the contaminants and the reactive packing materials. The wall thickness is thus a function of groundwater seepage velocity and the rate of reaction between the contaminant and the media. For a continuous PRB trenched system, the wall should extend deep enough into the aquifer, so that it will intercept the whole contaminant plume passing through the location of the wall. The underflow of contaminated groundwater should not occur as the hydraulic conductivity of the wall is higher than that of the aquifer. For the funnel-and-gate system, the funnel is preferably keyed into an underlying impermeable layer to prevent contaminant underflow.

8.5.3. *Reactive Materials*

Reactive materials are selected for their ability to reduce the target contaminants present in the groundwater by transformation into an innocuous state or by immobilization, so that the discharge stream complies with remediation concentration goals. Besides being capable of removing contaminants, the medium should be low cost, easy to obtain, devoid of hazardous materials, long lasting, and made of uniform particle size to prevent plugging and to allow model predictions to be accurate. The particle size should be optimum; large enough to allow unrestricted groundwater flow but small enough to give large surface area for desired reactivity or reaction rate that is dependent on available surface area.[34,35]

There are various types of reactive materials that can be placed in the reactive barrier, depending on the specific type of contaminants to be removed, the desired removal processes, and the ability of the materials in cleaning up the target contaminants. The media may degrade, adsorb, precipitate, or remove contaminants including chlorinated solvents, metals, nonmetallic inorganic chemicals, and radionuclides. The media that have been in use include zerovalent iron (Fe^0), amorphous ferric oxide, steel wool, copper wool, activated carbon, zeolite, catalyzed hydrogen, oxygen, phosphate, limestone, and sodium dithionite.[36] Fe^0 in the form of Fe chips, granular iron, or colloidal powder is by far the most extensively used reactive material in PRBs. The inexpensive Fe^0 is a mild reducing agent and readily available. Adsorbents are less commonly used in PRBs as compared to reducing agents, with GAC as the most common among adsorbent media used in sorption barriers.

8.5.4. *Reactions Mechanisms Involving Treatable Contaminants and Fe^0*

Fe^0 has been extensively used in PRBs to treat chlorinated organic compounds and inorganic contaminants.[34] In addition to the contaminants listed in Table 3, Fe^0 has potential to degrade nitroaromatics and pesticides.[37–39]

8.5.4.1. Chlorinated organic compounds

Fe^0 reacts with chlorinated solvents through reductive dechlorination to produce nontoxic and easily biodegradable daughter products such as light aliphatic hydrocarbon compounds, chloride (Cl$^-$), iron ions, and iron hydroxides. The reductive dechlorination reaction can be viewed as the coupling anodic and cathodic reactions occurring at the iron metal surface:

Anodic (oxidation) reaction:

$$\mathrm{Fe^0 \rightarrow Fe^{2+} + 2e^-} \tag{6}$$

Cathodic (reduction) reaction:

$$\mathrm{RCl + 2e^- + H^+ \rightarrow RH + Cl^-} \tag{7}$$

Net reaction:

$$\mathrm{Fe^0 + RCl + H_2O \rightarrow Fe^{2+} + RH + OH^- + Cl^-} \tag{8}$$

Redox transformations of chlorinated hydrocarbons are generally rapid, following either first-order or pseudo-first-order kinetics. The dechlorination rates vary for the various chlorinated solvents. A number of studies have shown that the degradation rate is proportional to the specific surface area or the surface area of iron per unit volume of water.[35,40–42] The half-lives for iron-induced dechlorination of various chlorinated aliphatic hydrocarbons are reported as ranging from less than a day to several days.[34] As a rule of thumb, perhalogenated hydrocarbons tend to be reduced more rapidly than their less halogenated congeners, and dechlorination is more rapid at saturated carbon centers (e.g., carbon tetrachloride) than at unsaturated carbon centers (e.g., TCE or VC).[34] Feng and Lim found that Fe^0 has ability to transform carbon tetrachloride and chloroform into a nonchlorinated end product, i.e., methane.[43]

Figure 18 illustrates the reductive dechlorination of TCE to ethene and ethane (which can be mineralized by biodegradation) through two competing but interconnected degradation pathways: (a) sequential hydrogenolysis and (b) reductive β-elimination.[44] The sequential hydrogenolysis pathway results in the production of DCE and VC as the intermediate products, which are slower to degrade than is TCE

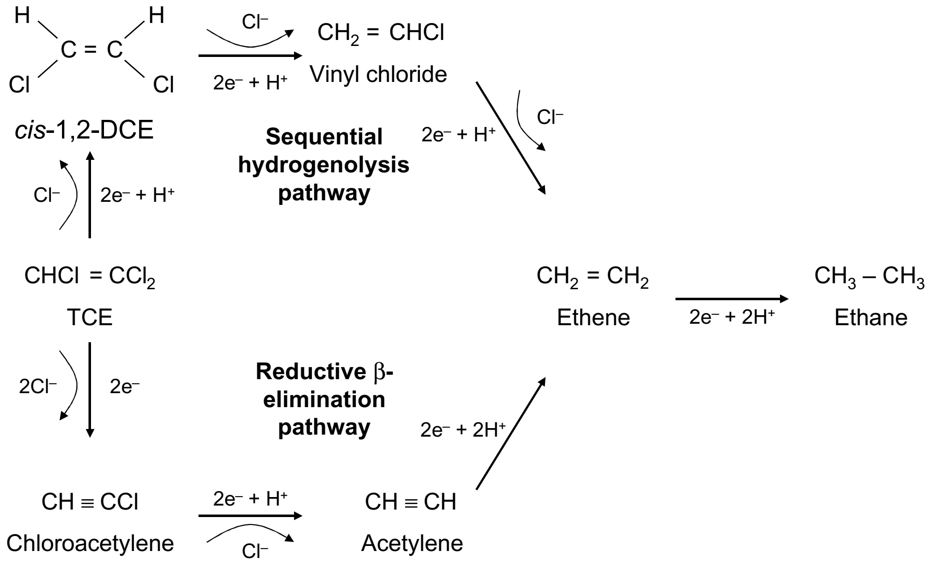

Figure 18. Reductive Dechlorination of TCE to Ethane and Ethane by Zerovalent Iron.[34,43]

itself. In contrast, the *β*-elimination pathway produces chloroacetylene, which is a very short-lived intermediate and is very rapidly reduced to ethene. VC is more hazardous and more persistent than TCE. Therefore, a PRB must be designed to provide adequate contaminant residence time in the treatment zone for the parent compound and its intermediate products to be fully degraded to nontoxic products. For a site where the groundwater is contaminated with a mixture of chlorinated hydrocarbons, the least reactive constituent usually determines the design of the PRB.

8.5.4.2. Redox-sensitive inorganics

Fe^0 has also been used to remove inorganic contaminants including Cr(VI), As(III), As(V), Se(VI), and Mo(VI) from aqueous solution via reductive precipitation of the oxyanion species of these elements[45–48] except As(V) and As(III). Reduction of chromate (CrO_4^{2-}) by Fe^0 has been extensively investigated and demonstrated.[45,49,50] Comparison of reaction rates observed using Fe^0, FeS_2, and $FeCO_3$ indicates that more rapid Cr(VI) reduction is attained using Fe^0.[45,51] The removal of Cr(VI) from solution is accompanied by a sharp decrease in the Eh, from initially oxidized conditions (Eh $> 100\,\text{mV}$) to very reduced conditions (Eh $< -300\,\text{mV}$), and a sharp increase in the pH from initially near neutral conditions ($6.5 < \text{pH} < 8.5$) to more basic conditions (pH > 9.5).[34] The Cr(III) formed is removed from the solution via the formation of insoluble Cr(III) hydroxide precipitates. The overall reactions for the reduction of Cr(VI) by Fe^0 and the subsequent precipitation of Cr(III) and Fe(III) oxyhydroxides are[34]:

$$CrO_4^{2-} + Fe^0 + 8H^+ \rightarrow Fe^{3+} + Cr^{3+} + 4H_2O \tag{9}$$

$$(1 - x)Fe^{3+} + (x)Cr^{3+} + 2H_2O \rightarrow Fe(1 - x)Cr_xOOH(s) + 3H^+ \tag{10}$$

For both As(V) and As(III), surface precipitation or adsorption by the corrosion products of Fe^0, typically ferrous oxides and ferric oxides, are the predominant removal mechanisms.[52] The incoherent and porous nature of oxide films on Fe^0 may allow adequate and continuous adsorption of both As(V) and As(III) over years of operation in the subsurface. In recent years, Fe^0 has been found to have potential for treating nitrate contaminated groundwater. Nitrate reduction by Fe^0 is a spontaneous process occurring favorably under acidic conditions,[53–55] and the reaction can be represented by the following equation:[53]

$$NO_3^- + 4Fe^0 + 10H^+ \rightarrow NH_4^+ + 4Fe^{2+} + 3H_2O. \tag{11}$$

8.5.4.3. Interferences

Interference with contaminants–Fe^0 reactions can occur under several scenarios. If the groundwater is oxygenated, DO is usually the preferred electron acceptor (oxidizing agent) over the chlorinated hydrocarbon and it will trigger a series of reactions that lead to consumption of the Fe^0 media and precipitation of ferric hydroxide or (oxy)hydroxides:

$$2Fe^0 + O_2 + 2H_2O \rightarrow 2Fe^{2+} + 4OH^- \tag{12}$$

$$4Fe^{2+} + 4H^+ + O_2 \rightarrow 4Fe^{3+} + 2H_2O \tag{13}$$

$$Fe^{3+} + 3OH^- \rightarrow Fe(OH)_3(s). \tag{14}$$

Under anaerobic conditions, Fe^0 can be also consumed by water:

$$Fe^0 + 2H_2O \rightarrow Fe^{2+} + H_2 + 2OH^- \tag{15}$$

$$Fe^{2+} + 2OH^- \rightarrow Fe(OH)_2(s). \tag{16}$$

Both Eqs. (12) and (15) result in an increased pH in weakly buffered systems, yielding ferric (oxy)hydroxides in aerobic systems (14) and ferrous (oxy)hydroxides in anaerobic systems (16). Ferric (oxy)hydroxides can clog the media and thus reducing hydraulic permeability. Ferrous hydroxides will be converted to nonpassivating magnetite coatings[56] that likely do not restrict flow and seem to allow sufficient contaminant degradation rates that can be sustained over years of operation in the subsurface.[34] However, an increase in pH because of Fe corrosion under both aerobic and anaerobic environments can also cause carbonate precipitates and aggravate media clogging. Additionally, anaerobic systems with a large reservoir of Fe, a favorable pH, and an available substrate can promote the activity of sulfate reducing bacteria (SRB) and methanogens present in the subsurface. SRB can convert SO_4^{2-} to S^{2-} using Fe^0 as electron donor. This can result in a rapid corrosion of the barrier, a build-up of precipitates, and biofouling.[57]

8.5.5. *Monitoring*

Field monitoring of PRB performance is an integral part of PRB implementation. There are two monitoring programs carried out in a typical PRB system. Compliance monitoring is carried out to evaluate the efficacy of plume treatment and ensure acceptable downgradient water quality as defined by cleanup standards. Compliance monitoring parameters include the contaminants of interest, potential contaminant daughter products, as well as the routine water quality parameters such as pH, alkalinity, specific conductance, temperature, etc. Typical monitoring well locations would be immediately downgradient of the reactive zone discharge, at each end of the wall, below the wall, and above the reactive zone.

Performance monitoring is carried out to evaluate the long-term performance of a PRB system and the reactions taking place in the treatment zone in order to confirm or modify earlier prediction and fate and transport model. Through performance monitoring, changes in the monitored parameters signal potential problems with the barrier. For a reactive iron barrier, the performance monitoring parameters should include hydraulic parameters, hydrologic parameters (baseline and changes over time), precipitates and rate of buildup, as well as geochemical indicator parameters such as Eh, DO, and Fe^{2+}. Increases in the hydraulic head loss across the barrier suggest that clogging is occurring. The buildup of precipitates can be assessed by sampling and evaluating surfaces of the reactive materials. When functioning properly, Eh in the iron barrier should be low, DO should be undetectable, and Fe^{2+} should be present.

8.5.6. *Advantages and Disadvantages*

PRB is gaining popularity as an alternative groundwater treatment technology to conventional PAT system and containment system for a number of reasons:

- It is an *in-situ* remediation system, and the treatment walls are likely to remain effective for up to 30 years, with minimal worker oversight.
- It is a passive remediation system; its operation does not depend on any external labor or input energy and thus only requires limited maintenance and operational cost.
- No above-ground treatment facilities are required other than monitoring wells following installation; i.e., sites can be productively used during cleanup.
- No waste stream treatment and disposal are required.

- It does not alter the overall groundwater flow pattern as much as high-volume pumping introduced by PAT, and, therefore, it does not cause mixing of contaminated and uncontaminated groundwater.
- Contaminants are not brought to the surface; i.e. cross-media contamination and worker exposure to contaminants are minimal.

The notable shortcomings of PRBs are:

- Remediation is a lengthy process for aquifers with low hydraulic conductivity because PRB systems rely on the natural movement of contaminant plume.
- Contaminant plume must be very well characterized and delineated; otherwise, a PRB may not be able to capture the whole plume.
- Most contaminant sources are immobile; therefore, a PRB cannot treat the contaminant sources (e.g., NAPLs).
- Clogging from biological or chemical precipitates is not uncommon.
- There are potential construction complications if the above-ground structures and underground utilities exist.
- Current technology only allows for relatively shallow plume treatment up to approximately 15 m below ground surface.
- Initial expenditure for construction and installation of a PRB is higher than that of a PAT system.

8.6. *In-Situ Groundwater Bioremediation*

8.6.1. *Description*

In-situ bioremediation is a technology that uses microorganisms to remove contaminants at a contaminated site. Often, these organisms are indigenous to the area and may even be adapted for growth on the contaminants in that particular environment. The adapted microbes will transform the contaminants until a limiting factor for microbial metabolism appears, for example, insufficient nutrient supply. In enhanced *in-situ* bioremediation, the limiting nutrients (such as nitrogen and phosphorus) are added to the environment to allow the organisms to continue degrading the contaminants. If the contaminants are recalcitrant to indigenous microorganisms, nonnative microorganisms specially suited to degrading the contaminants will be introduced.

Bioremediation can effectively degrade a wide range of organic constituents that are dissolved in groundwater and adsorbed onto the aquifer matrix. For example, hydrocarbons (aliphatic hydrocarbons, gasoline, fuel oil, etc.), alcohols (including glycols and phenols), ketones, esters, chlorinated compounds, explosives, pesticides, and other organic compounds have the potential to be biodegraded. Biological reactions involved in the biotransformation of organic compounds are shown in

Table 4. Microbial Catalyzed Biochemical Transformation Reactions.[66,67]

Reactions	Examples
Substitution	
a. Solvolysis, hydrolysis	$CH_3CH_2CH_2Br + H_2O \rightarrow CH_3CH_2CH_2OH + HBr$
b. Other nucleophilic reaction	$CH_3CH_2Br + HS^- \rightarrow CH_3CH_2SH + Br^-$
Oxidation	
a. α-Hydroxylation	$CH_3CHCl_2 + H_2O \rightarrow CH_3CCl_2OH + 2H^+ + 2e^-$
b. Epoxidation	$CHClCCl_2 + H_2O \rightarrow CHClOCCl_2 + 2H^+ + 2e^-$
Reduction	
a. Hydrogenolysis	$CCl_4 + H^+ + 2e^- \rightarrow CHCl_3 + Cl^-$
b. Dihaloelimination	$CCl_3CCl_3 + 2e^- \rightarrow CCl_2CCl_2 + 2Cl^-$
c. Coupling	$2CCl_4 + 2e^- \rightarrow CCl_3CCl_3 + 2Cl^-$
Dehydrohalogenation	$CCl_3CH_3 \rightarrow CCl_2CH_2 + HCl$

Table 4. Many of these reactions are similar to abiotic transformations, except the reactions are catalyzed by enzymes. The biodegradability of organic constituents also depends on their chemical structures and physical/chemical properties. In general, highly water-soluble organic compounds with short chain and low molecular weight are degraded more rapidly and to lower residual levels than are long-chain, high-molecular-weight, and less-soluble constituents, with some exceptions (e.g., MTBE). Metals are difficult to treat by bioremediation, as they are elemental and, therefore, cannot be biodegraded. Despite these potential pitfalls, there has been significant progress recently in applying bioremediation technologies to treat metals.[58–61] Nonmetallic inorganic contaminants such as nitrate, perchlorate, and cyanide are also amenable to bioremediation.[62–65]

8.6.2. *Microbial Metabolism and Cometabolism*

Organic contaminants serve two purposes for the organisms: (1) they provide a source of carbon, which is one of the basic building blocks of new cell constituents, and (2) they provide electrons, which the organisms can extract to obtain energy. In the biodegradation of contaminants, microorganisms secrete enzymes to catalyze energy-producing chemical reactions that involve breaking chemical bonds and transferring electrons away from the contaminants. This is an oxidation–reduction reaction; the organic contaminants are oxidized while the chemical that gains the electrons is reduced. The contaminants are called the electron donor, while the electron recipient is called the electron acceptor. The electron acceptors preferably used by the microorganisms in respiration are in the following sequence: O_2, NO_3^-, Fe^{3+}, SO_4^{2-}, and CO_2.

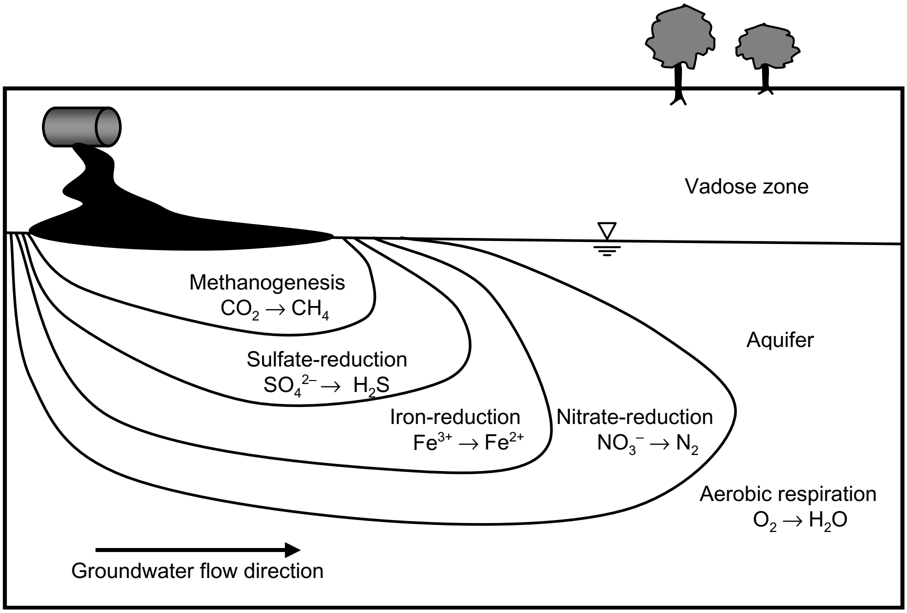

Figure 19. Sequence of Terminal Electron Acceptors Consumed Along the Flow Path of a Contaminant Plume in Groundwater.

In aerobic respiration, microbes use O_2 to oxidize the contaminants. In an environment with deficient DO, microorganisms that can exist without O_2 use anaerobic respiration to break down contaminants. In anaerobic respiration, NO_3^-, Fe^{3+}, SO_4^{2-}, and CO_2 can serve as electron acceptors. In addition to new cell matter, the byproducts of anaerobic respiration may include nitrogen gas (N_2), hydrogen sulfide (H_2S), reduced forms of metals, and methane (CH_4), depending on the electron acceptor. Figure 19 shows the preferential reductive sequence that involves the use of different electron acceptors throughout the contaminant plume. The ability of microorganisms to carry out a variety of respirations provides the opportunity to collect data during the site investigation phase that indicate whether or not a microbiological response to organic compounds has occurred. For example, if accurate measurements of DO in groundwater indicate that DO is present outside a plume of organic compounds but not detected within the plume, then a biological response may have occurred to consume DO. If methane or another respiratory end-product is detected within the plume, the results suggest that a biological response has occurred and that reducing conditions may exist.

A period of time may elapse before an increase in the rate of degradation is observed. This period of time is known as the acclimatization period, which varies and may range from less than 1 h to several months, depending on the chemical structure of the compound, subsurface biogeochemical conditions, and the concentration of the compound. Once acclimatization is achieved, the rate of degradation becomes a function of the processes controlling the availability of nutrients to the microorganisms and not of an intrinsic metabolic property of the microorganisms.

Sometimes enzymes produced for breaking down a specific compound (which serves as food substrates) may fortuitously degrade the less desirable compounds that are present as well. This process is termed cometabolism and is significant in the biodegradation of many hazardous organic compounds, especially the chlorinated solvents. Either direct metabolism or cometabolism can be enhanced in rate and extent with the availability of carbon and energy source, growth limiting nutrients, and suitable electron acceptors.

8.6.3. *Biodegradation Pathways*

Both aerobic and anaerobic biodegradations are important in bioremediation, with aerobic reactions being generally faster than anaerobic reactions. In general, oxidized organic contaminants such as chlorinated compounds are favorably reduced in anaerobic subsurface systems, while the reduced organic contaminants such as hydrocarbons are favorably oxidized under aerobic conditions. Petroleum hydrocarbons are most efficiently biodegraded under aerobic conditions due to their reduced nature. Among the readily biodegradable hydrocarbons are gasoline components such as alkanes, cycloalkanes, and BTEX. Figure 20 shows aerobic biodegradation of benzene. Creosote, which is a by-product of coal tar distillation for coke production, wood pyrolysis, and petroleum refining, is composed of a number of PAHs. The biodegradabilities of various PAHs decrease as their molecular weights (or the number of benzene ring) increases. Generally, the PAHs with three rings or

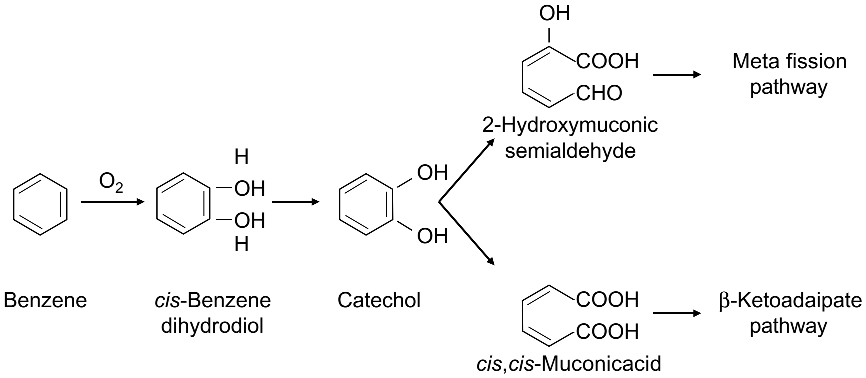

Figure 20. Pathway for the Aerobic Biodegradation of Benzene.[68,69]

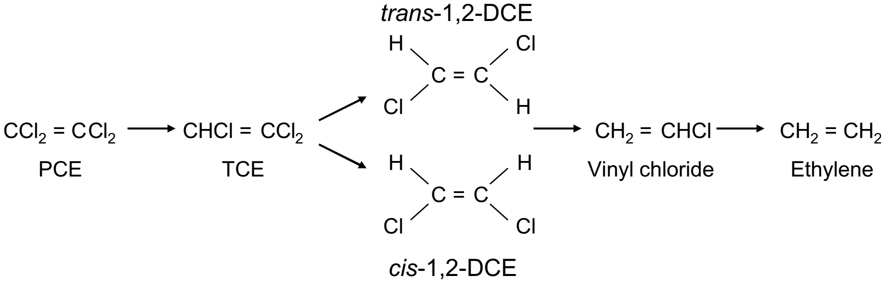

Figure 21. Anaerobic Reductive Dechlorination of PCE and TCE.[31]

less degrade at a greater rate than do the more complex PAHs. Many fungal species are capable of biodegrading PAHs under aerobic conditions.[70,71]

Chlorinated aliphatic compounds (such as chlorinated solvents) are generally more resistant to biodegradation. Biodegradation of these compounds may proceed by three metabolic pathways: aerobic metabolism, aerobic cometabolism, and reductive dechlorination via anaerobic processes. The pathway through which a chlorinated compound will degrade depends on the oxidation state of the contaminant and environmental conditions such as the DO and Eh levels. In general, the more reduced compounds proceed through oxidative metabolism and the oxidized compounds are degraded more favorably via reductive processes. Thus, the greater the number of chlorine or other halogens in the molecule is, the more degradable it will be in an anaerobic system because halogen can act as an electron acceptor in anoxic conditions. For example, PCE has been shown to be dechlorinated under environmental conditions that support the growth of anaerobic bacteria. This process is known as reductive dechlorination and is shown in Fig. 21. The compounds produced are TCE, the isomers of DCE, and VC. DCE has a greater potential for aerobic degradation than PCE does. However, the anaerobic reductive dechlorination of PCE and TCE leads to accumulation of VC, which is a known carcinogen.

Recent field and laboratory studies have discovered the occurrence of aerobic cometabolism of chlorinated solvents.[72–74] These cometabolic processes may lead to further degradation of DCE and VC produced because of anaerobic reductive dechlorination of PCE and TCE, causing their disappearance at the aerobic fringe of contaminant plumes.[75] This scenario is possible as contaminated groundwater containing DCE and VC migrates away from the anaerobic source area to an aerobic downstream region. Anaerobic biological treatment followed by aerobic biological treatment is now a proven technology that can remove chlorinated compounds from the environment in a cost-effective and environmentally acceptable manner.

Chlorinated aromatic compounds of frequent environmental concerns are chlorophenols, chlorobenzenes, PCBs, pesticides (e.g., DDT, 2,4-D, and 2,4,5-T),

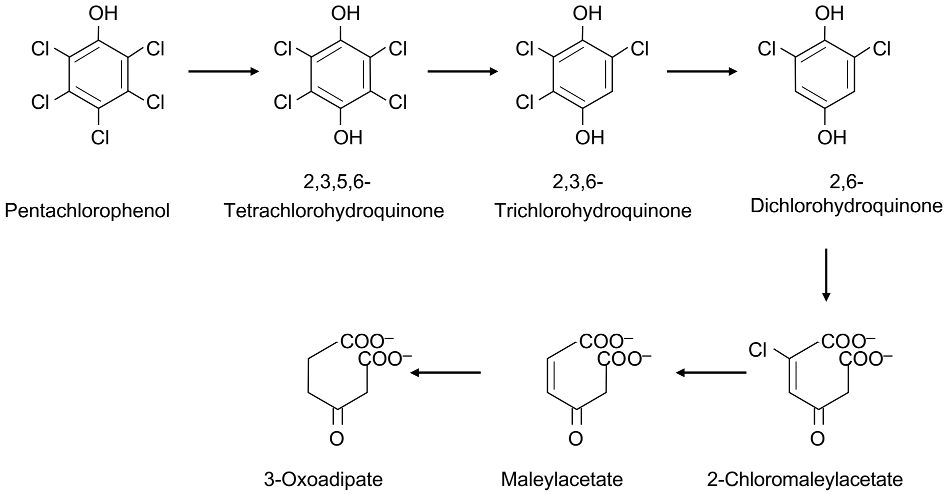

Figure 22. Pathway of PCP Biodegradation.
(*Source*: Modified from http://umbbd.ahc.umn.edu/pcp/pcp_map.html)

and pentachlorophenol (PCP), to name a few. The more chlorine the compound has, the more likely it undergoes reductive dechlorination by an anaerobic process. PCBs are most likely degraded by reductive dechlorination under anaerobic conditions, while aerobic degradation may be possible for PCBs with low number of Cl.[76] Biodegradation of DDT involves a combination of aerobic and anaerobic processes. The recognized first step in DDT biodegradation is anaerobic reductive dechlorination to DDD. PCP degradation pathway starts from monooxygenation of PCP to 2,3,5,6-tetrachlorohydroquinone, and subsequently follows reductive dechlorination to 2-6-dichlorohydroquinone (DiCH), as shown in Fig. 22. DiCH can be hydrolyzed and furthered reduced to 3-oxoadipate.

8.6.4. *System Design and Operation*

In-situ bioremediation can be implemented in a number of treatment modes, including aerobic, anoxic (nitrate respiration), anaerobic, and cometabolic. The goal of *in-situ* bioremediation is to stimulate indigenous microorganisms with nutrients, electron acceptors, and other chemicals that will enable them to destroy the contaminants according to the desired treatment modes. The ultimate consideration is when the targeted cleanup goal can be achieved.

The key parameters that determine the effectiveness of *in-situ* groundwater bioremediation are:

- Hydraulic conductivity of the aquifer, which controls the distribution of electron acceptors and nutrients in the subsurface.

- Biodegradability of the constituents, which determines both the rate and degree to which constituents will be degraded by microorganisms.
- The type of indigenous microorganisms.
- Physical and chemical forms of contaminants; whether they are dissolved in groundwater or adsorbed within the aquifer.

In a typical *in-situ* bioremediation system, groundwater is extracted using extraction wells to achieve hydraulic control over the plume to minimize spread of contaminants into uncontaminated areas. If necessary, the extracted groundwater is treated to remove residual contaminants. The treated groundwater is then mixed with an electron acceptor and nutrients, and other constituents if required, and reinjected upgradient of or within the treatment area using injection wells (as illustrated in Fig. 23) or infiltration galleries. A nutrient supply may not be necessary if the groundwater contains adequate amounts of nutrients.

To date, the most common *in-situ* bioremediation system in operation is intrinsic bioremediation, which relies mostly on indigenous microorganisms, stimulating them to work by supplying them with the optimum levels of nutrients and other chemicals essential for their metabolism. Bioaugmentation, a bioremediation system involving nonnative microorganisms, are used increasingly for site remediation.

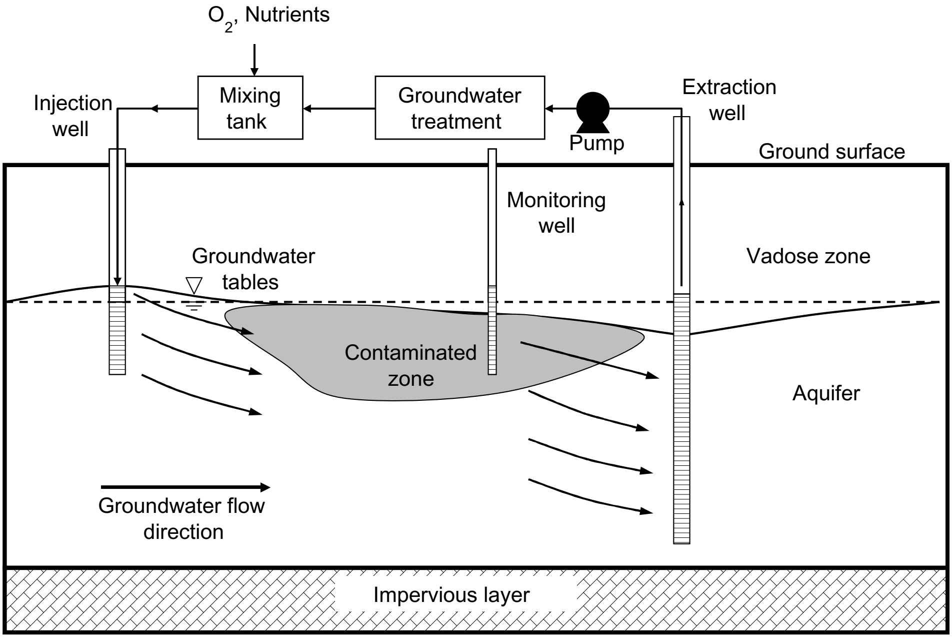

Figure 23. Typical *in-situ* Groundwater Bioremediation System Using Injection Well System for Nutrients and Oxygen Supplies.

In this system, the contaminated aquifer is augmented with nonnative microbes including genetically engineered microorganisms specially suited to degrading the contaminants of concern. Augmented bioremediation has the potential to enhance the bioremediation rate. Therefore, it may be chosen over intrinsic bioremediation where time allowed for contaminant elimination is short. It is possible that bioaugmentation will widen the range of treatable contaminants for future bioremediation systems.

Intrinsic bioremediation, bioaugmentation, a combination of the two, or a mixture of bioremediation with nonbiological treatment strategies, can be implemented to treat the groundwater contaminant plume and the source area. Contaminant concentrations in groundwater plumes are typically much lower than in the source area. Recoverable free product should be removed from the subsurface before operation of *in-situ* groundwater bioremediation system. This will mitigate the major source of contaminants as well as reduce the potential for smearing or spreading high concentrations of contaminants.

8.6.5. *Advantages and Disadvantages*

The advantages and disadvantages of *in-situ* bioremediation are listed in the following sections.

8.6.5.1. Advantages

- Contaminants that are dissolved in groundwater as well as adsorbed onto or trapped within the geologic materials can be removed.
- It is considered minimally intrusive and usually less disruptive of facility operations and infrastructure compared to other remediation technologies.
- It is generally recognized as being less costly than other remedial options.
- It can be combined with other technologies (e.g., bioventing, SVE) to enhance site remediation.
- In many cases, this technique does not produce waste products that must be disposed of.

8.6.5.2. Disadvantages

- Injection wells or infiltration galleries may become plugged by microbial growth or mineral precipitation.
- High concentrations of contaminants (e.g., TPH greater than 50,000 ppm) may be toxic to microbes.
- The technology may have difficulty in remediating low-permeability aquifers.
- Continuous monitoring and maintenance are often required to ensure effective degradation of the target contaminants and no harmful by-products remained.

8.7. *Other Groundwater Cleanup Technologies*

8.7.1. *Monitored Natural Attenuation*

Monitored natural attenuation (MNA) is an *in-situ* remediation technology that relies on naturally occurring processes in soil and groundwater that act without human intervention to reduce the mass, toxicity, mobility, volume, or concentration of contaminants in those media. The important components of natural attenuation are biodegradation, chemical transformation, sorption, dispersion, dilution, and volatilization (or evaporation). In order to implement MNA, it must be assured that natural remedial processes will be protective of both the public health and environment and achieve remedial goals within a reasonable time frame.

Consideration of this option often requires very detailed site investigations, detailed understanding of site biogeochemistry, modeling, and evaluation of contaminant degradation rates and pathways. The site investigation should be able to indicate the location and concentration of the contaminants, and the possible modes of contaminant transport in the subsurface. Biogeochemical processes likely to occur in subsurface, and the levels of associated compounds such as O_2, NO_3^-, Fe^{3+}, SO_4^{2-}, CO_2, and nutrients should be assessed to predict the possible contaminant transformation processes. The primary objective of site modeling is to demonstrate that natural processes of contaminant degradation will reduce contaminant concentrations below regulatory standards before potential exposure pathways are completed. Therefore, a strict long-term site-monitoring plan must be instituted to confirm that contaminant degradation is proceeding at rates and following pathways that meet cleanup objectives. Figure 24 shows a typical MNA scenario.

The major advantages of MNA include a lower cost than most active remedial alternatives and minimal disturbance to the site operations. MNA is increasingly used in the United States in the Superfund, Resource Conservation and Recovery Act (RCRA), and UST programs. The Superfund program, for example, used MNA as a remedy at more than 25% of contaminated groundwater sites.[77] There have been concerns about the broad use and, perhaps, misuse of MNA. Some view this technique as a “do-nothing” approach. There are skepticisms about the ability of MNA in achieving cleanup goals within a reasonable timeframe and migration of constituents to receptors may occur within this period. MNA is deemed ineffective where constituent concentrations are high (e.g., TPH higher than 20,000 ppm), and especially heavier petroleum products and recalcitrant compounds are the main constituents. MNA is also unsuitable if an impact on groundwater supply and surface water bodies is expected. Therefore, MNA adoption does not preclude the ultimate need to actively clean up contaminated sites. The U.S. EPA expects that MNA will be most appropriate when used in conjunction with other remediation measures such as source control and groundwater extraction, or as a follow up to active

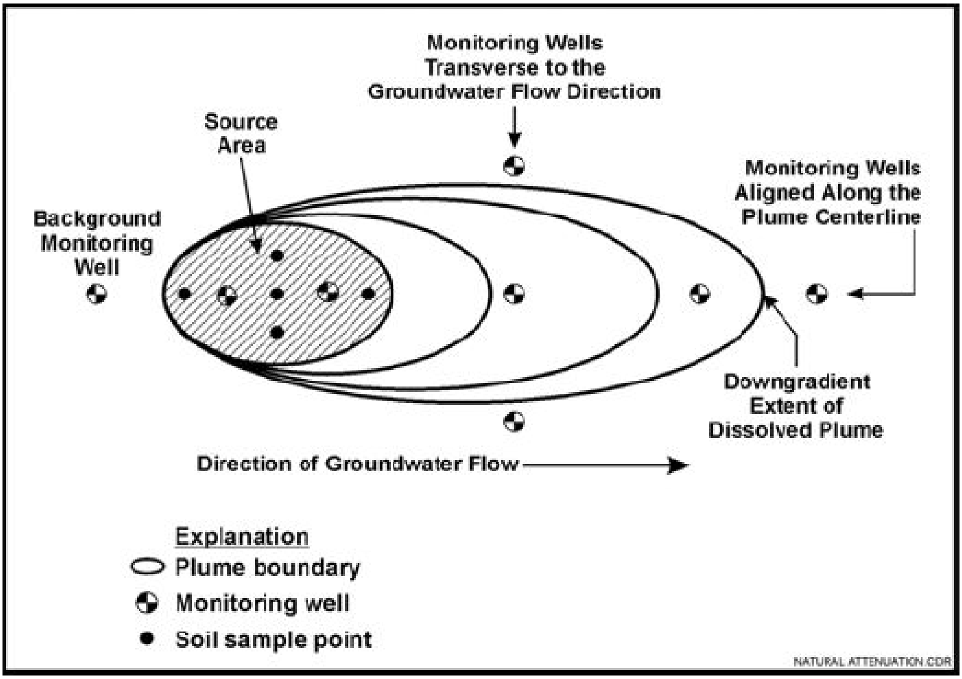

Figure 24. Diagram of MNA of a Contaminant Plume and the Locations of Monitoring Wells and Sampling Points.[29]

remediation measures that have already been implemented (Directive 9200.4-17P, U.S. EPA).

8.7.2. *Phytoremediation*

Phytoremediation is a process that uses vegetation to clean up contaminated soil and groundwater. There are several mechanisms through which phytoremediation takes place, as shown in Fig. 25. The mechanisms are:

- *Phytoaccumulation* — The uptake and storage of contaminants in the plants stem or leaves. Some plants, called hyperaccumulators, draw contaminants through the roots and allow the contaminants to accumulate in their stems and leaves many folds above normal limits.
- *Phytovolatilization* — The uptake and vaporization of contaminants by a plant. This mechanism draws contaminants and transforms them to an airborne vapor. The vapor can either be the pure contaminant, or the contaminant can be metabolized by the plant before it is vaporized, as in the case of mercury, lead, and selenium.[78–80]

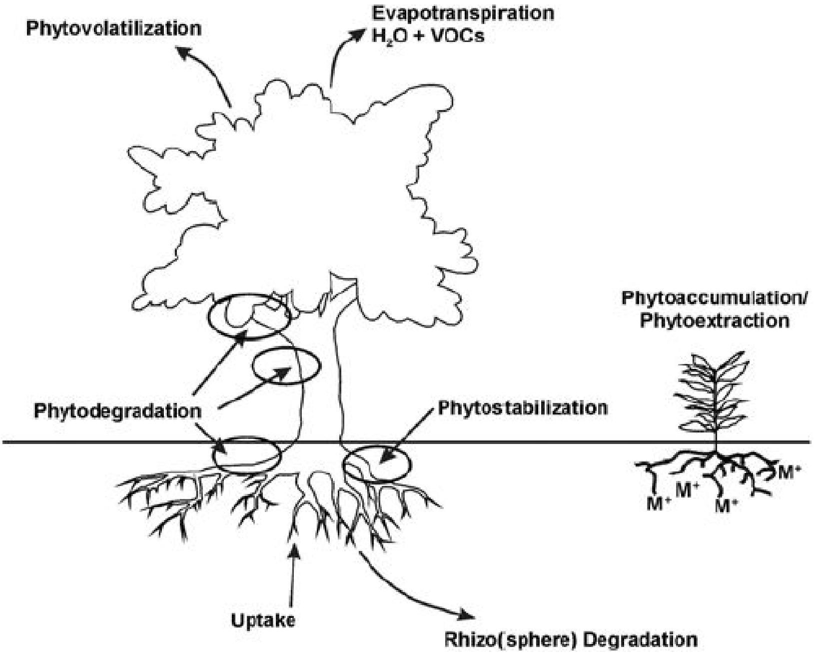

Figure 25. Conceptual Diagram Showing Phytoremediation System and the Possible Mechanisms Through that Contaminants are Removed.[29]

- *Phytodegradation* — The process of uptaking, assimilating, and then degrading contaminants by plants using the plant-derived enzymes such as nitroredictase, laccase, dehalogenase, and nitrilase. This process has been demonstrated in field studies.[78] The daughter compounds can be either volatilized or stored in the plant.
- *Phytostabilization* — Contaminants are immobilized at the interface of roots and soil through production of chemical compounds by plants.

Some plants may display any combination of the above phytoremediation mechanisms when they are grown at the contaminated sites. The most effective phytoremediation currently in practice combines these four mechanisms to clean up a site. For example, poplar trees can accumulate, degrade, and volatilize the contaminants in the remediation of organics.[81]

Groundwater phytoremediation systems are designed to control or contain contaminated groundwater using deep-rooted and water-loving trees. A stand of deep-rooted trees can act as a "biological pump", removing a considerable amount of water from the saturated zone. During the growing season, when the trees are actively transpiring, a zone of capture is created, in which all of the groundwater within a specific thickness of the saturated zone is used by the trees. As plants take up contaminated water via transpiration, contaminants may become adsorbed to root tissue or soil particles, biodegraded in the rhizosphere, or taken up by the plant. Another practice of groundwater phytoremediation is through effective construction of wetlands in cleaning up the groundwater.[82]

Phytoremediation offers several advantages over traditional cleanup technologies. It is passive, solar driven, low cost, and has wide public acceptance. Phytoremediation systems do not generate noise. The practice improves the esthetic quality of the sites, which can be used as recreational facilities, such as parks or golf courses, or wildlife habitats. Having plants covering the contaminated ground reduces wind and water erosions and reduces risk of human exposure to contaminants. Phytoremediation can also yield marketable resources, such as pulp and paper. However, phytoremediation has some limitations. It is usually not an immediate solution, as plants sometimes require several growing seasons to reach maturity. The technology may also be restricted by weather patterns and plant predation, diseases, or phytotoxicity. Trees usually have roots of 3–4.5 m long and, therefore, cannot remediate deep aquifers. Phytoremediation may pose another risk as toxic substances absorbed by the plants may enter the human food chain. Phytovolatilization of contaminants may lead to air pollution hazard. Since plants only absorb water-soluble substances, phytoremediation is inefficient for contaminants that bind tightly to soil.

8.7.3. *Hydraulic and Pneumatic Fracturings*

Hydraulic fracturing uses pressurized water (injected through wells) to create cracks in contaminated low-permeability and over-consolidated soils or sediments and, therefore, enhance their permeability. The fractures created in the process are filled with a porous medium such as sand or special materials that can facilitate extraction efficacy, delivery of substrates and nutrients for *in-situ* bioremediation, or pumping efficacy. Typical applications of hydraulic fracturing are linked with SVE, *in-situ* bioremediation, and PAT systems.

On the other hand, pneumatic fracturing uses highly pressured air to fracture contaminated consolidated soil and sediments to extend existing fractures and create a secondary network of fissures and channels. This enhanced fracture network increases the permeability of the materials to liquids and vapors and accelerates the removal of contaminants, particularly by vapor extraction, biodegradation, and thermal treatment. The gases released are then captured and treated above ground, as illustrated in Fig. 26.

One of the shortcomings of this technology is that fractures created by hydraulic and pneumatic fracturings will close or collapse due to overburden pressure in nonclay soils. Another shortcoming is inability to control the final location or size of the fractures that are created, and this may lead to the creation of new pathways

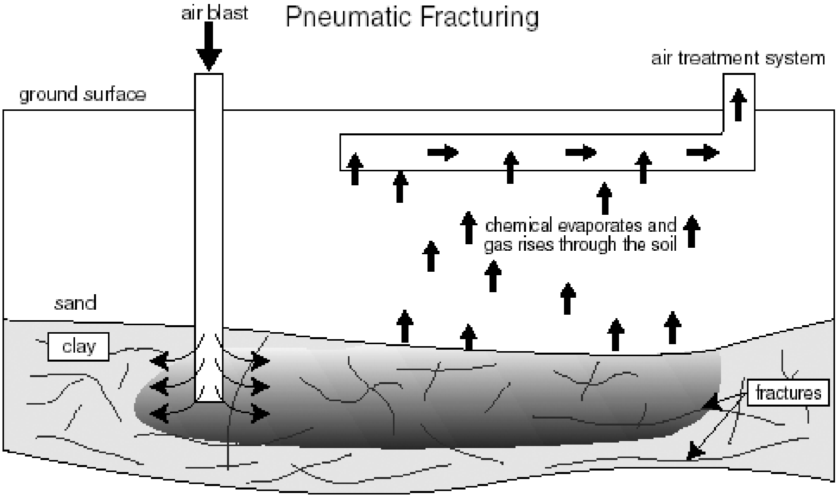

Figure 26. Conceptual Diagram of Extending Fractures and Creating Fissures via Pneumatic Fracturing.[83]

leading to the unwanted spread of contaminants (e.g., DNAPLs). Special care should be taken when it is applied in subsurface with underground utilities, structures, or entrapped free product. The technology should not be used in bedrock susceptible to seismic activity.

8.7.4. *In-Situ Thermal Treatment*

In an *in-situ* thermal treatment process, heat is introduced into an aquifer to vaporize volatile and semi-volatile contaminants. The vaporized components as well as the groundwater are removed by vacuum extraction for treatment above ground, as shown in Fig. 27. Thermal remediation methods are effective because at elevated temperature, the mass transfer of contaminants through volatility, evaporation, and diffusion increases. In addition, the rates of biodegradation and most chemical reactions also increase with temperature. A rise in temperature will also decrease the viscosity of NAPLs and water, and thus facilitate mass transfer.

In-situ thermal treatment has a limited history of full-scale application, compared to *in-situ* AS and SVE. A variety of application methods commercially available are *in-situ* steam heating, hot air injection, *in-situ* thermal desorption, six-phase electrical heating, three-phase electrical heating, direct conduction from heaters, and microwave heating. Steam or hot air injection is commonly applied to treat aquifer and vadose zone, while the others are applied to treat vadose zone though they may also be applied to groundwater.

The *in-situ* thermal treatment is able to greatly accelerate remediation process and significantly reduce the cost. However, the treatment method involves

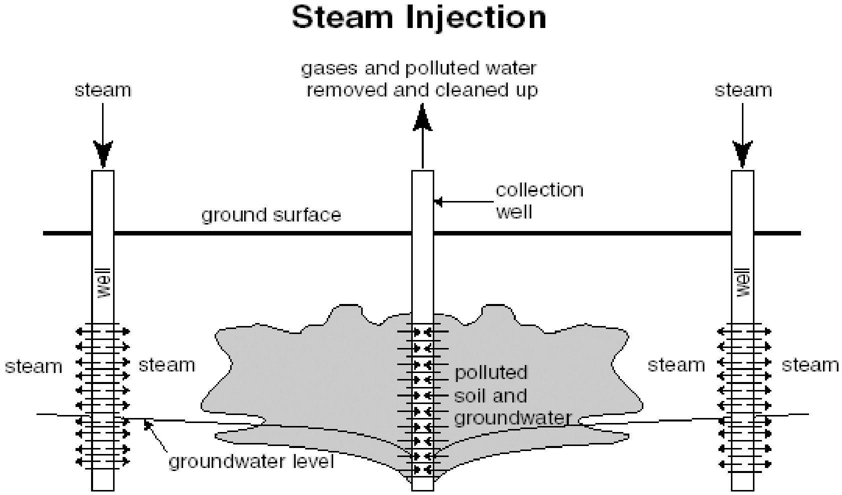

Figure 27. Conceptual Diagram of *in-situ* Thermal Treatment of a Contaminant Plume.[84]

significant capital expenditures. Therefore, the thermal process is normally short- to medium-term technologies. Compare to *in-situ* AS, all the approaches to *in-situ* thermal treatment are more intensive, typically resulting in a cleanup duration in the range of 8–12 weeks. The initial cost to mobilize and set up equipment for *in-situ* thermal treatment typically is greater than that for AS and, even with the shorter operating duration, the overall cost of heating technology typically is higher than the cost for AS. *In-situ* thermal treatment typically would be selected in favor of AS only in cases where rapid remediation is essential or there is significant quantity of free-phase DNAPL distributed as ganglia throughout the saturated zone.[29]

8.7.5. *In-Situ Chemical Oxidation*

In-situ chemical oxidation (ISCO) is based on the delivery of chemical oxidants to contaminated media in order to destroy the contaminants by converting them to innocuous compounds. The oxidants applied in this process are typically hydrogen peroxide (H_2O_2), potassium permanganate ($KMnO_4$), sodium persulfate ($Na_2S_2O_8$), ozone (O_3), Fenton's reagent, or, to a lesser extent, chlorine and DO. These oxidants have varying strengths of attraction for electrons, and thus have varying degrees of effectiveness on common contaminants. Hydrogen peroxide can be costly, and because of its volatility requires preservative measures. Nevertheless, the shorter treatment process associated with H_2O_2 may save on labor and operating costs. Potassium permanganate is a stable and easily handled oxidant in both solid and solution forms. The discovery of $Na_2S_2O_8$ as a potential oxidant for ISCO application has been relatively new, and it has been reported by Liang

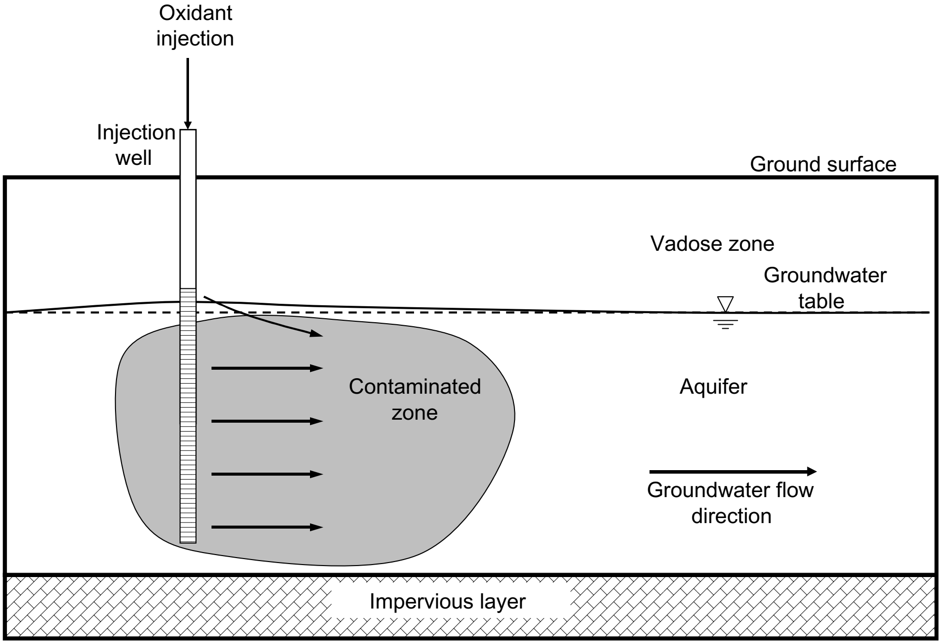

Figure 28. Conceptual Diagram of Field-Scale Application of ISCO.

and his coworkers.[85,86] A field study reported by Zhu *et al.* showed that chlorine dioxide with small percentages of Cl_2, O_3, and H_2O_2 were effective for remediation of a petroleum-contaminated aquifer.[87] Enrichment with DO has been more often associated with enhanced bioremediation. The most common field applications of ISCO thus far have been based on Fenton's reagent whereby H_2O_2 is applied with an iron catalyst creating a hydroxyl free radical. This hydroxyl free radical is capable of oxidizing complex organic compounds. Residual H_2O_2 decomposes into water and oxygen in the subsurface and any remaining iron precipitates out. Lim and Goh observed that Fenton's process may be also triggered with the application of H_2O_2 alone if the soil (or aquifer) contains sufficient iron oxides.[88]

The methods for delivery of the oxidants may vary. Figure 28 shows a scheme whereby an oxidant is injected upgradient or into the contaminated area through injection well systems. If the oxidant is applied at optimum level sufficient for complete oxidation of contaminants, no extraction well system is needed to extract and recirculate the oxidant-laden groundwater.

ISCO can be applied in conjunction with other treatments such as PAT and SVE to break down remaining compounds. It may be used as an alternative to *in-situ* bioremediation due to its ability to oxidize compounds that are not readily biodegradable and may be also used in concert with *in-situ* bioremediation by the oxidation of the recalcitrant compounds and allowing biodegradation of the daughter products. ISCO may be used in applications where the effectiveness of bioremediation is limited by the range of contaminants and/or climate conditions. ISCO has been used to treat VOCs including DCE, TCE, PCE, and BTEX as well as SVOCs including pesticides, PAHs, and PCBs. In addition, it can be used to treat an arsenic plume by immobilizing the dissolved arsenic through oxidation of the more mobile As(III) to the less mobile As(V).

ISCO treats both dissolved and sorbed contaminants concurrently and the treatment time is significantly shorter compared with PAT remediation. The cost of treatment is also lower than that of PAT remediation. There is no requirement for above-groundwater treatment or disposal. However, the range of applicable sites for ISCO is more limited than for AS. It may be difficult to achieve good mixing between the groundwater and the oxidant solution. Successful application of ISCO is very sensitive to site conditions such as natural organic matter content and hydrogeology. Sometimes, the injected solution tends to react with natural organic matter before it reacts with the contaminants.

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Chapter 15

ENVIRONMENTAL MONITORING OF NEARSHORE DREDGED MATERIAL OCEAN DISPOSAL SITES

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Abstract
In Canada, annual monitoring programs are conducted at selected dredged material ocean disposal sites to meet Canada's treaty obligations under the *London Convention 1972* and its *1996 Protocol*. Results of these monitoring activities are used to confirm the appropriateness of decisions allowing dredged material disposal at sea and to assess the disposal impacts. This chapter introduces and describes the methods and tools used by Environment Canada for monitoring of nearshore dredged material ocean disposal sites.

Keywords: Dedged material, ocean disposal site, physical monitoring, sediment chemistry and toxicity tests, benthic community survey.

1. Introduction

Dredging is required to remove sediments from shipping channels and harbors for the deepening of marine ports and navigable waterways. Dredged materials removed from the shipping channels usually consist of sand and silt with low or undetectable contaminant levels while those from the harbors are generally composed of silt and clay with higher levels of contaminants.[1–4] In some cases, dredged material is used for beneficial purposes to create artificial islands for wildlife habitat, for beach nourishment, or to serve as construction materials for shoreline maintenance.[5–8] However, dredged material is often disposed of in nearshore marine, estuarine, and harbor waters due to comparative economic and environmental advantages over land disposal options.

Aquatic disposal of dredged material will produce both physical and chemical changes that may result in changes in the benthic community. There is also potential to interfere with the functional uses of the water body. The predominant physical effects are related to changes of turbidity, suspended solids concentrations, sedimentation rates, as well as burial of organisms. Disposal of contaminated dredged material can cause short- and long-term chemical and biological effects.

The effects of dredged material disposal include those related to offsite transport of material away from the disposal site, e.g. smothering of aquatic organisms, alteration of fisheries and wildlife habitat, and interference with other uses of the area such as fishing and recreation. Controlled dumping with effective mitigation measures can reduce the transport of fines and localize damage. If dumping at a disposal site is terminated for several years and if only uncontaminated materials were disposed of at the site, the recovery of the benthic communities at the site will occur.[9–12] The long-term impacts of dredged material disposal are more of a concern if the dredged materials are contaminated and if the disposal site receives dredged materials annually over a long time period.

The disposal of dredged material in marine waters of a country is controlled by its own regulatory framework. For many countries (e.g. Canada, the United States, the United Kingdom, France, Germany, Russia, Japan, and China), regulatory requirements reflect commitments made by way of international conventions, one of which, the *London Convention 1972* (LC 72) — is global in scope. The *London Convention* applies to all marine waters other than the internal waters of a country. In 1996, the convention was modernized with the resulting changes — the *1996 Protocol* — implemented in 2005. Contracting parties (i.e. signatories to the LC 72) must apply the requirements of the protocol to marine waters. For internal marine waters, countries may either apply the protocol or adopt other effective permitting and regulatory measures. Under Annex 1 of the LC 72, dredged material is considered one of the categories of waste that may be considered for disposal at sea.[13–16]

To confirm the appropriateness of a decision allowing dredged material disposal at sea and to refine the assessment practices of disposal at sea permit system, contracting parties conduct monitoring programs at selected disposal sites using the dredged material assessment framework (DMAF) that defines monitoring as measurement of:

- Compliance with permit requirements and
- Condition (and changes in condition) of the receiving area to assess the impact hypothesis upon which the permit was approved

The LC 72 also requires contracting parties to submit annually the disposal site-monitoring reports to the Secretary-General of International Maritime Organization.[16]

In the United States, the US Army Corps of Engineers (USACE) have done extensive monitoring studies at dredged material ocean disposal sites. One of the most active programs is the disposal area monitoring system (DAMOS) that is administrated by the New England District of USACE (DAMOS Contribution Series).

In Canada, dredged material disposal site monitoring is also an essential component of Environment Canada's Disposal at Sea Program. Guidance documents have been developed by Environment Canada for conducting monitoring programs at dredged material ocean disposal site to ensure a high quality of information collection and national consistency.[17,18] Most of the monitoring activities in Canada are jointly conducted by Environment Canada and Natural Resources Canada, and sometimes with the participation of the Department of Fisheries and Oceans. Integrated techniques used in these studies include acoustic surveys (multibeam, side-scan sonar, and sub-bottom profiler surveys); current meters; seabed erosion measurement; sediment chemistry; sediment toxicity; benthic biota sampling; and bottom photographs and video imaging.

This chapter will focus mainly on the technologies and methods used by Environment Canada and its partners for dredged material disposal site monitoring.

Where it is appropriate, innovative techniques and methods used in the United States will be introduced.

2. Triggers to Monitor Dredged Material Disposal Sites and Site Selection

The Environment Canada National Guidelines for Monitoring Dredged and Excavated Material at Ocean Disposal Sites identifies three criteria to assist in identifying the need to monitor a disposal site[18]:

- A permit for the open-water disposal of dredged material was issued following the failure of the dredged material to meet chemical screening criteria but subsequent conformity with biological testing criteria.
- There are possible effects on nearby sensitive areas, including habitats, or potential conflicts with other nearby uses of the sea.
- The volume of material disposed in any one year is greater than 100,000 cubic meters.

The guidance also recommends that major sites should be monitored on a cycle of no more than five years.

In conformity to the Environment Canada national guidelines, the Environment Canada Atlantic Regional Office adopted a dredged material disposal site matrix selection table to rank their regional disposal sites in order of priority. The matrix used a set of criteria to provide scores to each disposal site and rank the sites in order of priority for monitoring. For each selected site, a review is conducted to gather historical information and data collected from the site. The information and data are interpreted, summarized, and an impact hypothesis is developed to define the level and effort of monitoring activities to be conducted at the site.

The Environment Canada national monitoring guidance also recommends using a tiered monitoring approach to test the impact hypotheses to ensure that site monitoring is conducted in a cost-effective and consistent fashion (Fig. 1). The first tier is to delineate the disposal site boundaries, measure accumulation of dredged material within and outside the boundaries, and document evidence of offsite transport of the dredged materials. The second tier involves chemical and biological assessments. This includes measuring the contaminant levels in the seabed materials within and/or outside the disposal site boundaries and conducting toxicity tests, bioaccumulation studies, and benthic community surveys. When potential human health or environmental concerns are detected in the second tier, further monitoring will be conducted under the third tier to investigate the long-term use of the site and the need for site remediation or closure.

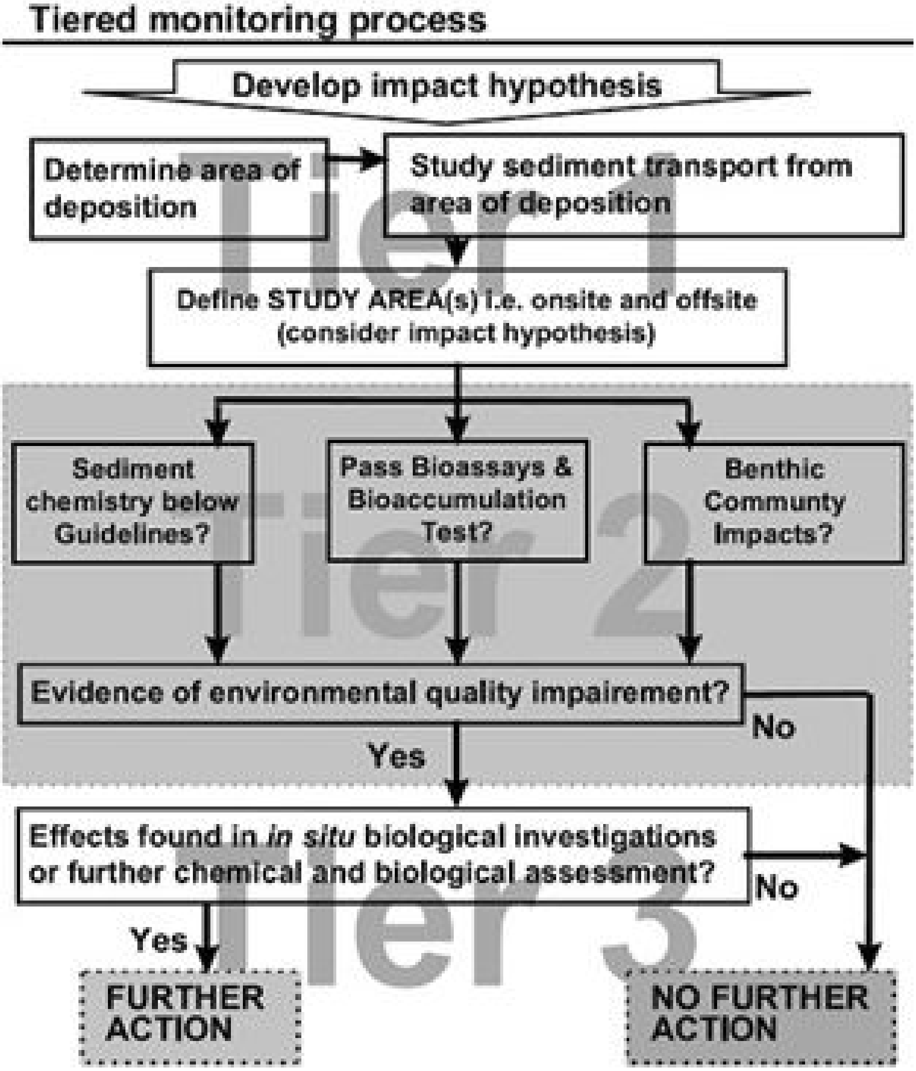


Figure 1. Tiered Monitoring Process for Ocean Disposal Sites.

3. Physical Monitoring

Physical monitoring is typically conducted in the first tier of a monitoring program to collect physical and geological data for disposal site delineation, site stability evaluation, and assessment of offsite sediment transport. This is the critical stage of a monitoring program. Data collected from this survey will be used for the design of further physical surveys and chemical and biological monitoring in the next tiers.

Physical monitoring generally involves the use of an integrated survey using a combination of geophysical, sedimentological, and geochemical techniques to characterize the nature, distribution, and remobilization of marine sediments at the disposal sites and its vicinity.[19,20] These surveys provide information on bathymetry and the character of seafloor sediments at the disposal sites and in adjacent areas. Geophysical characterization and mapping of the seafloor sediments are accomplished using acoustic geophysical methods including sidescan sonar, multibeam bathymetry, and sub-bottom profiler surveys. Seafloor sediment samples, photographs, and video transects are generally taken during this phase of the monitoring to ground truth the geophysical surveys and to provide information of the physical characteristics of the sediments and related benthic habitat. The sediment samples are analyzed to provide information on the grain size and the organic and inorganic geochemistry of the material. The photographs and underwater video transects provide additional information of the character and composition of the sediments on the seafloor. All visible benthic fauna on the pictures and video are identified to the lowest taxonomic level possible and presence of each recorded. Organisms identified during the analysis of the bottom photographs are logged with a representative photo of each. The presence of burrows, trails, and tracks of motile fauna are described as references to the possibility of infauna or epifauna present.

Geophysical samples and bathymetry data for the disposal area are generally compiled as part of the first tier of a monitoring program and used to interpret changes in the character of the disposal site. All data are integrated into a geographic information system (GIS) to show the positions of new and existing samples, photographs, analyses, etc. in relationship to multibeam bathymetry images, sidescan sonar, and sub-bottom profiler data. The use of geographically referenced datasets enables detailed study of the spatial and temporal relationships at the disposal site.

3.1. *Acoustic Geophysical Surveys*

3.1.1. *Multibeam Bathymetry*

Recent developments in multibeam bathymetry systems have resulted in improvements in the resolution and calibration of these systems.[21,22] This has lead to a dramatic change in the way that disposal site-monitoring surveys are performed. Environment Canada and Natural Resources Canada use multibeam bathymetric survey techniques as the primary reconnaissance mapping tool to survey all dredged material disposal sites (Fig. 2).[19,20] The technique provides a rapid means of determining the morphology and nature of the sediments on the seafloor bottom. Instead of using conventional single-beam bathymetric equipment, which covers only a small percentage of the total seafloor area, multibeam bathymetric equipment uses a multi-element transducer to provide many individual soundings of the water depth and echo

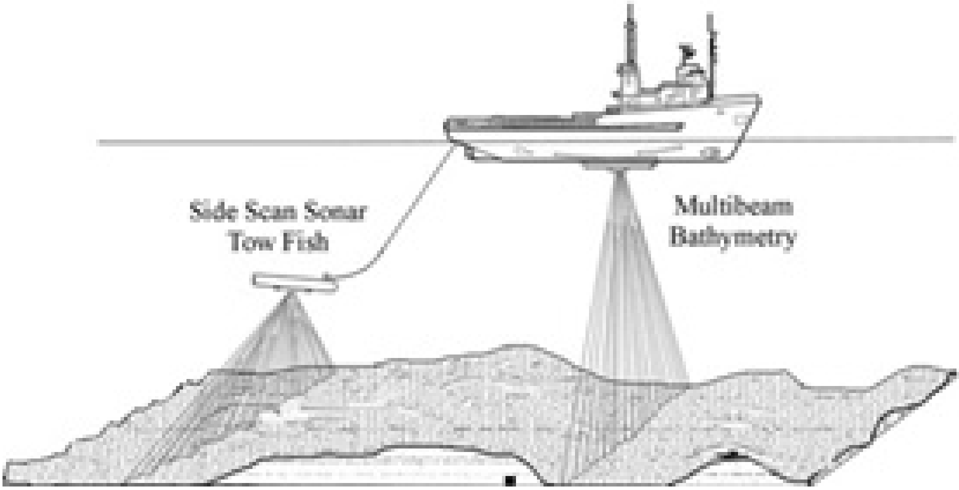


Figure 2. Side Scan Sonar and Multibeam Bathymetry Techniques used for Disposal Site Surveys.

strength across a swath of the seafloor. As shown in Fig. 2, the systems use an array of acoustic beams to provide dense sounding data both along and between track lines. They provide accurate and high-resolution bathymetric data of the seafloor over relatively large areas.

A wide swath (up to seven times the water depth or greater) can be surveyed in a single pass through an area. Survey lines are spaced to ensure that swaths from adjacent lines overlap and provide complete coverage of the seafloor. This redundant coverage increases the reliability of the bathymetric data, but also increases the volume of data generated. In order to deal with the large quantity of data, automatic seafloor tracking programs determine depths and echo strengths for each transducer element, correct for transducer motion, and calculate a geographic co-ordinate for each individual sounding. Data are processed to remove spurious soundings and poorly referenced navigation data, and to correct for variations in the velocity of sound in the water column and differences in tidal height. The processed bathymetry data are imported into a GIS where shaded-color relief images are generated and overlaid on bathymetry charts of the area to accurately depict the morphology of the disposal sites and surrounding seafloor.[19–22] The features such as channels, areas of bedrock outcrop, as well as small-scale features including small bedforms, rocks, individual dredge spoils, anchor drags, and debris are readily identified on the images. The maps and images form the basis for the preliminary interpretation of geological processes and morphological features and are used to plan subsequent surveys using higher-resolution sidescan sonar and seismic reflection systems to better define features identified on the multibeam data. The improved resolution of multibeam bathymetry techniques has allowed detection of features that are not always detected by ordinary bathymetric soundings.

Most multibeam systems can simultaneously measure the strength of the reflected sound, called acoustic backscatter intensity. Acoustic backscatter intensity values are controlled by the physical properties of the seafloor sediments such as the velocity of sound and the density and roughness of the sediment. Backscatter generally increases as the sediments on the seafloor become denser and less porous, and increase in grain size. Hard acoustic returns indicate rock, gravel, or coarse sand while softer returns indicate mud, silt, or fine sand. Mapping the distribution of backscatter provides valuable information on the character and distribution of coarse and fine sediments within an area. A considerable amount of research has gone into techniques to classify sediments based on the acoustic backscatter response.[23] The technology allows the production and rendering of detailed 3D images of the seabed showing geological and disposal site features in considerable detail.

A wide variety of multibeam bathymetry systems are available, ranging from portable systems that can be temporally mounted on available vessels, to permanent installations using hull-mounted transducers and dedicated vessels. All systems required use of accurate attitude sensors to allow processing to remove vessel motion from the data, and a method of correcting for changing vessel elevation due to changes in tides.

3.1.2. *Sidescan Sonar*

Sidescan sonar systems use acoustic pulses to obtain high-resolution images of large areas of the seafloor, without all on the infrastructure required for multibeam bathymetry surveys. Many sidescan sonar systems are designed to be lightweight, easy to transport to a survey area, and quick to mobilize on vessels of opportunity. A team of one or two people can generally install a basic system on a vessel in a couple of hours. All components for shallow water surveys are designed to be hand carried by one or two personnel. Lightweight cables and towfish allow the system to be deployed and recovered by hand, without the use of winches. More robust and heavier systems are available for deep water surveys requiring the use of heavy cables and winches.

A towfish, deployed near the seafloor, uses side-looking transducers to send out acoustic pulses across the seabed. Sound that is reflected off the bottom or off objects on the seafloor is then received by the transducers and displayed as continuous scans to create detailed images along a swath of the seafloor (Fig. 2).[24] These images provide information about the topography and texture of the water–sediment interface, based on the acoustic backscatter characteristics of the sediment. Objects with a strong acoustic impedance contrast and high roughness, such as rocks, coarse sand, gravel, and some types of debris, will produce high acoustic backscatter intensities. Zones of fine-grained sediments and smooth surface, such as fine sand, silt, and mud produce lower acoustic returns. Changes in the relief of the interface will also produce echoes from the surface insonified by the sonar, controlled by the angle of incidence between the signal and the surface. Acoustic shadows will appear when areas are masked by the zones of higher relief.

A survey is planned to run a set of survey lines with predesigned spacing to provide full coverage of the seafloor. Data collected from adjacent survey lines are then combined to produce a mosaic of the sidescan sonar data and produce an image of the entire surveyed area.

The width of the swath surveyed is a function of the frequency of the sidescan sonar system. Lower-frequency systems (around 100 kHz) provide wide swath coverage and are used to create mosaics of a large area. Higher-frequency systems (300 kHz and above) can provide higher resolution images but have shorter ranges and are generally used to image a particular feature or area of interest. Many sidescan sonar systems provide dual-frequency operation to provide both wide coverage and high resolution. These data reveal detailed information of distinct objects or features on the seafloor. Some sidescan sonar systems are quite sensitive and can measure features on the ocean bottom smaller than 10 cm.

3.1.3. *Sub-Bottom Profiler*

Sub-bottom profiling systems use a technique that is similar to that employed by echosounders and medical ultrasounds, but use lower-frequency sound pulses that penetrate the sediments and reflect from the interfaces between sedimentary layers to provide vertical cross sections of the seabed. The echoes map changes in the acoustic properties of the sediments, specifically the acoustic impedance (Z), which is a function of the density (ρ) in kg/m^3 of the material and the velocity (c) in m/s at which sound travels through the material.[25]

For normally incident waves,[23] the acoustic impedance can be calculated as

$$Z = \rho c. \tag{1}$$

The reflection coefficient R between layers can be calculated as:

$$R = \frac{\rho_2 c_2 - \rho_1 c_1}{\rho_2 c_2 + \rho_1 c_1}. \tag{2}$$

The acoustic reflectivity of adjacent layers depends on the contrast in the acoustic impedance on opposite sides of an interface boundary. The higher the contrast in acoustic impedance between adjacent layers is, the more energy is reflected. Note that there would be no reflection between adjacent layers if the acoustic impedance for the two layers were equal, in spite of differences in velocity and density. If an underlying layer has a lower impedance than the overlying layer, the reflected wave

Table 1. Velocity, Density, and Calculated Reflection Coefficient Between Water and a Soft Sediment such as a Clay and a Harder Sediment such as a Sand.

Material	Velocity (m/s)	Density (kg/m^3)	Reflection Coefficient with Water
Water	1500	1027	
Soft sediment	1600	1200	.10
Hard sediment	1800	1800	.35

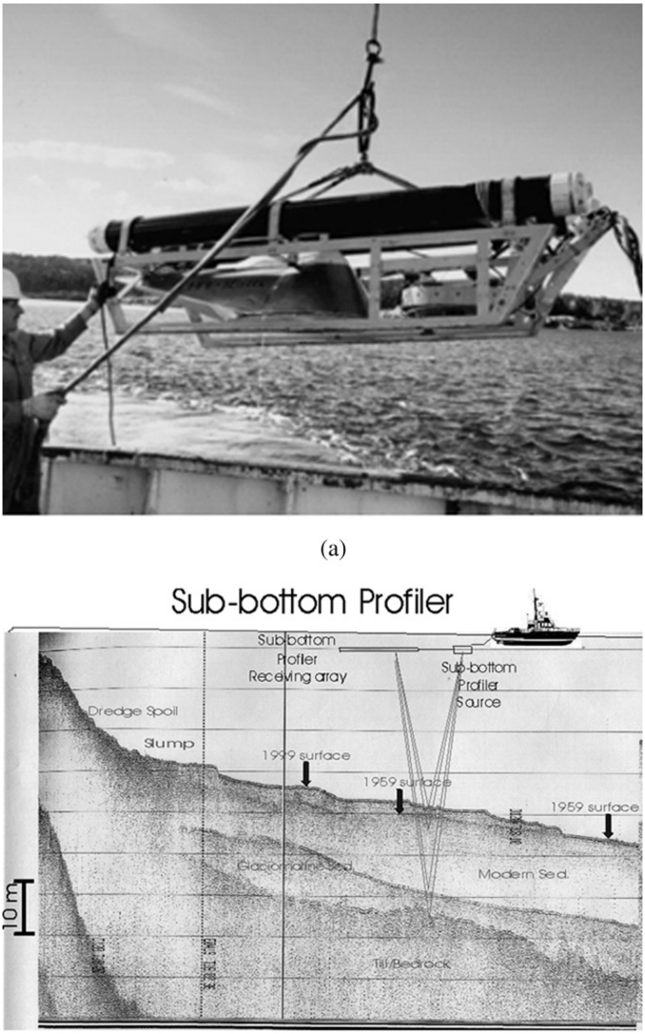
is phase inverted compared to the incident wave. This condition often occurs when there is gas entrained in sediments, or when there is a soft low-velocity layer, such as clay, overlain by a denser and higher-velocity sediment, such as sand.

Table 1 shows some theoretical values for the velocity and density of marine sediments. The reflection coefficient between the water column and an underlying soft sediment such as a clay is seen to be about 0.1; and about 0.3 for an interface between water and a harder sediment such as a sand.

The sub-bottom profiler systems typically generate acoustic pulses in the 1–10 kHz range. The pulses radiate outwards until there is a change in acoustic impedance, such as the water–sediment interface. A part of the transmitted sound is reflected back to the source, while the remaining sound energy penetrates through the boundary and into the sediments. The acoustic energy continues to penetrate the sediments until it encounters another contrast in acoustic impedance, when a part of the energy is again reflected. As the ship transits along the survey line, a series of these pulses are displayed on a graphic recorder or video screen to create a profile or cross section of the sediment layers beneath the sediment–water interface (Fig. 3). The profiles are used to determine the type, distribution, and thickness of sediments below the sediment–water interface, and provide information on the depositional history and sedimentation processes of sediment layers over time.

A variety of acoustic sources are used for sub-bottom profiling. Pingers are basically low-frequency echo sounders that operate at a single frequency and provide penetration into the sediments. Swept frequency sources such as "chirp" systems use a pulse that sweep over a range of frequencies. Matched filter processing techniques are then applied to remove the long wavelength of the signal associated with the chirp process and to improve the resolution and signal to noise resolution of the data. Electrodynamic systems (boomers) discharge high voltage through a spiral coil to produce rapid movement of a nearby plate to produce a short-duration broad-frequency band acoustic pulse. A discussion of the various systems can be found in Lurton.[26]

Many of the disposal sites monitored by Environment Canada and Natural Resources Canada are located in the areas that have been transgressed by glaciers



(b)

Figure 3. (a) Seistec® Towed Catamaran Sub-Bottom Profiler. (b) Graphic Record Obtained with Seistec® Sub-Bottom Profiler Over the Black Point Disposal Site Showing How Typical Acoustic Pulses Travel Through the Water and are Reflected Back to the Receiver. An Interpretation of the Sediments is Provided.

during the last 10,000 years. The sediments frequently contain the areas of highly stratified glaciomarine sediments underlain by glacial till. In many nearshore areas, these sediments have been transgressed by rising sea levels, resulting in winnowing of the local sediments to produce lag surfaces composed of sands and gravel. The fine-grained components are generally transported into basins in deeper water. This wide variety of sediment types require the use of a sub-bottom profiling system with high resolution that can still penetrate deep into the seafloor sediments. Environment Canada and Natural Resources Canada have been using surface-towed and deep-towed electrodynamic (boomer) sound sources in the majority of the recent monitoring surveys. These systems provide resolution of 0.25 m or better and have the penetration required to profile through thick accumulations of dredge spoils.

3.2. *Currents and Sediment Transport Surveys*

Two key factors control the stability of dredged materials on seabed: the shear stress exerted on the sediments by waves or currents, and the erodibility of the sediments. Erosion of sediment occurs whenever the shear stress exerted on the bed by waves or currents exceeds a threshold value, which is known as the critical erosion shear stress. In order to predict if sediments will remain in place after placement at a disposal site, it is necessary to determine both the nature of the forces acting on the sediments and the ability of the sediments to withstand these forces. The measurement of water velocity from currents and waves can be performed with current meters. The measurement of the potential erodibility of sediments requires the use of computer models and specialized equipment such as *in-situ* flume tanks.

3.2.1. *Current Meters and Suspended Particulate Measurements*

As part of monitoring programs at disposal sites, currents and suspended particulate matter (SPM) have been measured by seafloor mounted equipment including current meters and acoustic and optical backscatter sensors (OBSs) deployed at selected locations in the vicinity of disposal sites. These systems provide information of the flow conditions at the deployment site and can be used to study the effect of storm events as well as normal tidal currents on the site.

Multi-sensor platforms such as RALPH have been developed by Natural Resources Canada and deployed at several locations to provide long-duration monitoring of currents and tides while taking time-lapse images of the seafloor to study the detailed relationship between current velocity and sediment transport (Fig. 4).[27] RALPH is an autonomous and bottom mounted instrumentation platform for long-term *in situ* measurements of waves, currents, and sediment transport processes in marine environments. The system can be configured with a variety of sensors and the sampling sequence programmed to provide data required for a wide range of studies. On a typical deployment, the main sensors on RALPH will include a pressure transducer, an array of four electromagnetic current meters (EMCM) at the heights of 10, 30, 50, and 100 cm above the seabed, and six OBS at heights of 10, 30, 50, 70, 100, and 150 cm above the seabed. RALPH can be programmed to burst sample for 15 min duration every hour at 5-Hz frequency, or other sequences as required for the monitoring project.

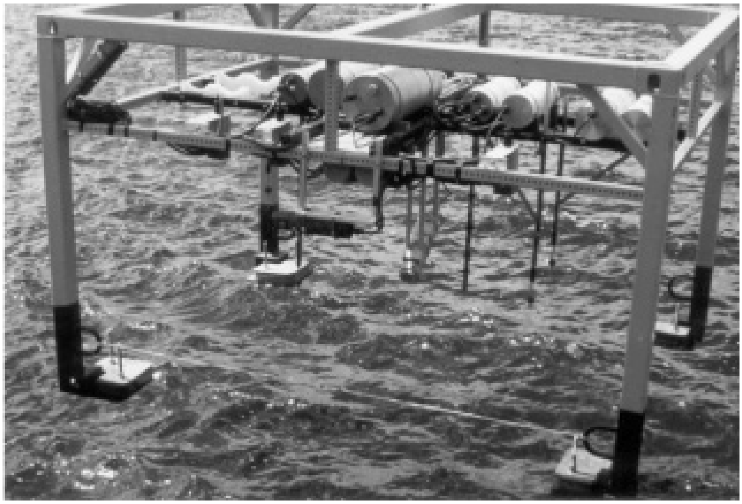


Figure 4. The Geological Survey of Canada Developed the **RALPH** Instrument for the Study of Seabed Dynamics. The Platform is Equipped with a Pressure Sensor, Current Meters, Sonar Technology to Measure Bedform Changes, and a Time Lapse Camera. It Also Uses Both Optics and Ultrasonics to Measure Suspended Sediment using a Digital Video Camcorder. Data is Stored on Internal Hard Disk Drives.

Acoustic Doppler Current Profilers (ADCP) have been used at several locations to monitor currents throughout the water column to provide information on the potential for dispersion of fine-grained sediments from dredge spoils as they descend through the water column. High-frequency echo sounders and towed backscatter sensors, used in conjunction with ADCP measurements, provided information on suspended particles.

3.2.2. *Sediment Transport Survey*

3.2.2.1. Benthic annular flume

Erosion of sediment occurs whenever the shear stress exerted on the bed by waves or currents exceeds a threshold value, known as the critical erosion shear stress. The ability of sediments to withstand erosion is controlled by several factors including the presence of fine sediments, which will result in cohesive sediment bed behavior. This behavior is a function of multiple parameters, including grain size distribution, bulk density, mineralogy, organic content, and chemistry. The cohesive bed behavior can best be determined by performing erosion experiments on the sediment of interest.

The Benthic Annular Flume, Sea Carousel, was developed by the Natural Resources Canada to measure *in-situ* marine sediment erosion rates (Fig. 5).[28,29] Sea Carousel is a tethered and *in-situ* annular flume, with an outer diameter of

Figure 5. The Sea Carousel is a Benthic Annular Flume that is Used to Study the Effects on Increasing Current Flow on Sea-Floor Sediments.

2.0 m, a width of 15 cm, and a height of 30 cm. Flow is driven by a rotating lid. OBSs provide information on suspended sediment concentrations. Sampling ports located in the outer channel wall allow the withdrawal of fluid for the measurements of SPM, particulate organic matter (POM), and chlorophyll *a*. The flume is lowered to the seafloor and allowed to settle for 30 min. Lid rotation is initiated and the rate increased incrementally every 5 min to measure the onset and duration of erosion.

Amos *et al.*[28] used the erosion threshold and erosion rate to characterize sediment erodibility and describe the stability of surface sediments below the upper layer. A high erodibility corresponds to a less stable sediment with a lower critical erosion shear stress and greater erosion rate. It is difficult to predict these parameters based on the physical properties of the sediment due to a wide variety of biological, geological, and physical factors that affect sediments during and subsequent to sediment deposition. Physical factors that affect the erodibility include sediment grain-size, bulk density, water content, air exposure, rainfall, and consolidation of intertidal flat sediments. Erodibility can be substantially modified by biological factors such as biostabilization or destabilization by various benthic macrofauna species or microphytobenthos.

Other systems including Sedflume,[30] a flume specifically designed to measure cohesive and mixed sediment erosion rates and the variation of these rates with depth using core samples. Flume experiments provide direct measurement of the character of the sediment under various flow conditions to determine if the sediment bed behaves in a cohesive or noncohesive manner, and are used to develop site-specific algorithms for erosion rate as a function of applied shear stress and the variation of these rates with depth. These algorithms are critical for model predictions of storm and ambient condition mound erosion and migration.

3.2.2.2. Artificial fluorescent sediment tracers

Artificial fluorescent sediment tracers can be used to monitor sediment transport direction and magnitude for various particle size classes.[31] Environmentally safe fluorescent tracer particles are manufactured to specific sizes and densities (sand and fines) and placed on or within the dredged material mounds. The tracer particles are then transported similarly to the dredged material. Sequences of surface grab samples are collected from multiple locations within the monitoring area and tested for fluorescence to detect the tracer material. Sample stations are reoccupied during the monitoring period to quantify movement of the sediment tracer with time. Sediment tracers can be manufactured with distinct fluorescent signatures to allow tracking of multiple sediment sizes (sand and fines) and sources and distinguish both source and grain size for transported material. These data can then be used to estimate the amount of fine and coarse material from several disposal areas and the direction and magnitude of transport from the disposal site.

4. Chemical and Biological Monitoring

4.1. *Chemical Monitoring*

Chemical monitoring is conducted within and/or outside of the disposal site boundaries to assess the levels of contamination in dredged material. This is especially important if the dredged material originates from dredging activities in major harbors or near a water body receiving land-based sources of discharges.

Disposal permits are generally issued for the disposal of dredged materials containing low levels of contaminants. To confirm the low levels of contaminants, chemical monitoring programs are conducted at selected disposal sites to ensure compliance. For contaminated dredged material, a disposal permit is issued only if the results of an assessment indicate that the potential environmental risks of disposing of these materials at the disposal site can be eliminated or greatly reduced by following strict environmental plans that control the dredging and disposal activities and the placement and isolation of the materials at the site.

If the results of the chemical monitoring program indicate contaminants in the dredged materials at the disposal site are at concentrations that pose a potential risk to the environment or human health and there is evidence that the contaminants may be moving offsite to the surrounding areas, then a more extensive chemical monitoring program and an associated biological monitoring program would be needed to further assess the significance of environmental effects resulting from use of the site.

A wide range of sampling grabs and corers are used for collecting sediment samples for chemical monitoring. The most commonly used grab samplers and corers include Ekman, Ponar, Van Veen, and Shipek grab samplers and Benthos gravity and Box corers.[32,33] In general, grab samplers are used for surface sample collection while corers are used at sites where subsurface samples are important for the monitoring programs. Samples are used for grain size and chemical analyses that normally include a suit of chemicals of concern and total organic carbon (TOC). When performing chemical analyses on the sediment samples, it is important to ensure that suitable chemical extraction methods and marine sediment standard reference samples are used and standard quality control (QC) and quality assurance (QA) procedures are followed. Due to the high cost of chemical analyses, especially for organic compounds, LC 72 Contracting Parties have adopted different suites of chemicals of concern for their dredged material assessment and monitoring programs. In Canada, the suite of chemicals of concern recommended by Environment Canada for disposal permits include several heavy metals such as Cd, Hg, Cu, Zn, and Pb and organic compounds such as polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs). When required, tributyltins (TBTs), dioxins, and furans are also added to the list of analytical parameters.[18]

Sampling design used for collecting sediment samples for chemical analyses at disposal sites is generally site specific. There are guidelines developed in Canada[34] and the United States[35] for collecting samples in different types of environment. In some cases, it is important to collect reference samples in the near vicinity of the disposal site or from another area with similar physical and chemical backgrounds.

In addition to assessing the levels of contamination at the disposal site, sediment chemical results can also be used to track the offsite movement of suspended and bottom sediments at the disposal site.[36]

4.2. *Biological Monitoring*

If the results of chemical monitoring program indicate that the contaminants in the sediments at a disposal site pose a potential risk to the marine organisms or human health and there is indication that the contaminants may be moving offsite to the surrounding areas, then a more extensive chemical monitoring program and an associated biological effects monitoring program would be needed to assess the potential environmental impacts.

Biological monitoring is a critical part of the dredged material monitoring program. Biological effects of dredged material disposal include both physical and chemical effects. Short-term physical impacts including direct damage and smothering of marine organisms may not be a major concern since it has been shown that the impacted benthic communities can recover within a few years if disposal activities are terminated at the site. However, when large quantities of dredged materials are disposed at a site repeatedly, the recovery of the benthic communities can be difficult. Often, the impacts can result in the establishment of a distinct benthic community at the site. If elevated levels of contaminants are detected at the same disposal site, the combined physical and chemical effects can be a long-term and complex environmental concern.

The most commonly used biological monitoring methods are the use of ecotoxicological tests (sediment toxicity and bioaccumulation tests) and *in situ* assessment of benthic community. These two methods are recommended in the Environment Canada dredged material disposal site-monitoring guidance.[18]

4.2.1. *Sediment Toxicity and Bioaccumulation Tests*

The biological testing methods for assessing toxicity and bioaccumulation of contaminants in marine sediments used in Canada and the USA are quite similar in the guidance documents published by Environment Canada and the US EPA.[37–42] Both countries use the same bioaccumulation guidance manual.[40]

Sediment samples for ecotoxicological testing are generally collected at the same time as samples are collected for chemical analyses or benthic community survey. Sediment samples are mixed immediately in clean container at the sampling sites. Thoroughly mixed samples are sub-sampled and stored at 4°C until used for the tests. It is recommended that samples stored for a maximum of 14 days before use, and must not be stored more than a maximum of six weeks.[37]

Biological tests used for dredged material disposal site monitoring commonly include a suite of lethal, sublethal, and bioaccumulation tests:

- A 10-day amphipod acute lethality test on whole sediment[37];
- a sublethal bacterial bioluminescence test (Microtox® solid-phase)[38];
- a sublethal echinoid fertilization test on sediment pore water[39];
- a sublethal 14-day growth and survival test with marine polychaetes on whole sediment[42]; and
- a bedded sediment bioaccumulation test on whole sediment,[40]

An effective biological monitoring for nearshore dredged material disposal sites program requires the combined use of a battery of appropriate toxicity tests and statistically based sampling designs. Using a battery of toxicity tests strengthens the evidence of toxicity in disposal site sediments. Since different species differ in their sensitivity, they might not even be responding to the same toxicants. This is important since disposal site sediments might contain a variety of contaminants not analyzed in chemical monitoring such as pesticides, pharmaceuticals, aliphatic hydrocarbons, asphaltines, wood preservatives, and certain antifouling compounds (e.g. TBT). In addition, it is important to recognize that some noncontaminant factors, such as grain size, TOC, sulfide, ammonia, salinity, and possibly other factors, can confound the interpretation of the sediment toxicity test results. Special care should be taken to eliminate these confounding factors when designing the monitoring program by selecting appropriate reference sites. The results of the sediment toxicity tests on disposal site sediments should be compared for statistically significant difference from a reference sediment.[43] In the absence of a suitable reference sediment, comparisons are made with a clean control sediment usually obtained from the site where the test organisms are collected or with sediment used to culture the test organisms.

There are many toxicity tests other than those discussed in this chapter that can be effective for assessing the toxicity of disposal site sediments using organisms such as bivalve mollusk embryos and bacterial communities.[44–50] To date, these sediment toxicity tests have not found wide use in the Canadian regulatory disposal site-monitoring program.

4.2.1.1. 10-day amphipod acute lethality test on whole sediment

Amphipod sediment toxicity tests have been used for the assessment of sediment contamination in Canada and the United States,[37,41] as well as in Europe, Asia, and Australia/New Zealand. Amphipods are sensitive to sediment contaminants, and their response to noncontaminant variables (grain size, ammonia) has been extensively studied. This allows for accurate interpretation of toxicity test results.

The 10-day amphipod acute lethality test on whole sediment is summarized in Table 2. The test species to be used for a particular study is selected based on tolerance to sediment variables such as grain size, interstitial salinity, and ammonia content.[37] Test animals can either be purchased from commercial suppliers, or collected locally, once their identification has been confirmed by a qualified taxonomist. Some species can be cultured in the laboratory for a year round supply of test organisms. A photograph of *Eohaustorius estuarius*, a species commonly used in Canada, which has a high tolerance to a wide range of salinities and grain sizes, is provided in Fig. 6.

The test is conducted according to the procedure recommended by Environment Canada.[37] Each container of test sediment (stored at 4°C) is homogenized and

Table 2. Summary of the Amphipod Acute Lethality Test Procedure.

Test organism	Marine and estuarine amphipods: *E. estuarius*, *Rhepoxynius abronius*, *Eohaustorius washingtonianus*, *Amphiporeia virginiana*, *Corophium volutator*, *Foxiphalus xiximius*, *Leptocheirus pinguis*
Type of test	10-Day static test on whole sediment; not fed during test.
Test vessels	1-L glass jars
Volume of test solution	175-mL sediment, 800-mL overlying water
# Test animals	20 per test container
Lighting	24-h continuous light
Test temperature	15±2.0°C (except 10±2.0°C for *A. virginiana*)
Experimental configuration	Five replicates per test sediment
Endpoints	Average % survival ± SD
Measurements	T, DO, pH, salinity, and ammonia in overlying water
Reference toxicants	$CdCl_2$ in a water only test

Figure 6. Amphipod *E. estuarius.*

175 mL portions added to each of five 1 L glass mason jars. The jars are then filled with 800 mL of clean seawater (salinity matched to interstitial water salinity for the test sediments), covered, then aerated overnight with oil-free compressed air at a rate of approximately 150 mL/min.

The following day, amphipods are removed from their holding sediment by sieving the contents through a 500-μm sieve. Animals are double-counted and 20 animals are added to each of the test vessels (Fig. 7). Testing is performed with a 24-h light photoperiod with lighting provided by overhead fluorescent fixtures at an intensity of 500–1000 lux. Test temperature is normally 15±1°C. The tests are

Figure 7. Adding Amphipods to Test Jars.

checked daily for observations, aeration, temperature, pH, salinity, ammonia, and dissolved oxygen.

After 10 days, the contents of each jar are sieved through a 500-μm sieve. Any immobile animals are observed under a microscope to determine mortality, defined as lack of all movement when observed under a dissecting microscope for 5–10 s. Any animals missing are assumed to be dead. The mean survival and standard deviation of each test sediment are calculated and compared for statistically significant differences.[43]

A reference toxicant test is conducted with cadmium chloride using a water only exposure for 96 h duration at the same time as the sediment test. Using the mortality data at each test concentration, the 96-h LC50 (concentration calculated to cause 50% mortality after 96-h exposure) is calculated using the methods of Stephan[51] with the Comprehensive Environmental Toxicology Information System (CETIS) statistical software.[52] The 96-h LC50 for this analysis is entered into the Shewhart Control Chart to ensure that the test is within standard operating limits, and that the population of amphipods used in the test is of normal sensitivity.

4.2.1.2. Bacterial bioluminescence test (microtox® solid-phase)

The solid-phase test for measuring sediment toxicity using the luminescent bacterium *Vibrio fischeri* has several advantages over the more conventional sediment tests used to assess the toxicity of contaminated sediments using macroinvertebrates:

- The solid-phase tests using the luminescent bacterium can be completed in hours as opposed to days for the other sediment tests.
- The small sample volume requirements mean ease of collection and shipping to the test laboratory.
- There is no time-consuming culturing of test organisms required, as the bacterial reagent (lyophilized *V. fischeri*) is available commercially.
- A single source of reagents and supplies leads to increased standardization and repeatability of results worldwide; validated by interlaboratory studies.[53]
- The test results have shown good correlation with contaminants, benthic community structure, and many conventional invertebrate whole sediment bioassays as discussed by Day *et al.*[54] and Zajdlik *et al.*[55]
- The same test can be conducted using freshwater, estuarine, and marine sediments, as well as on terrestrial soils.
- Bacteria form the basis of many important ecosystem functions such as biodegradation of organic matter and nutrient recycling and are an important group of organisms for inclusion in any battery of toxicity tests.

This test is conducted according to Environment Canada, 2002,[38] and uses a Microtox® M500 analyzer that features a photometer and an array of temperature-controlled incubator wells (Fig. 8). This method exposes the bacterium to the sediment sample; if toxic materials are present they interfere with the cellular

Figure 8. Microtox M500 Analyzer.

Table 3. Summary of the Microtox Solid-Phase Test Procedure.

Test organism	*Vibrio fischeri*, strain NRRL B-11177
Type of test	20 min static test on whole sediment
Test vessels	Disposable glass cuvettes
Volume of test solution	1.5 mL
Inoculum	20 μL reconstituted bacteria
Lighting	Does not apply
Test temperature	15±0.5°C
Experimental configuration	12 test concentrations and three control solutions
Endpoints	Moisture corrected IC50 in mg/L (dry-weight basis), others possible e.g. IC25
Measurements	Light levels of all test filtrates and controls measured
Reference toxicants	Toxic positive control sediment (e.g. certified reference sediment)

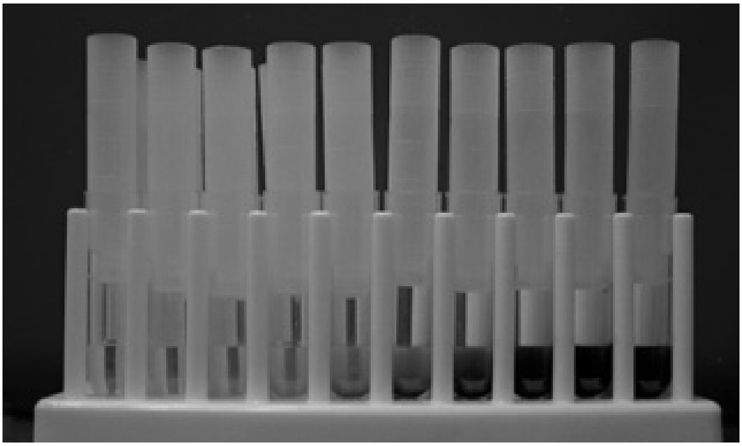


Figure 9. Preparation of a Series of Concentrations of Whole Sediment Sample by Serial Dilutions in Water Using a 50% Dilution Series.

respiration of the organism. This interference is measured as a decrease in light output by the bacterium, *V. fischeri.* The test is summarized in Table 3.

An aliquot of the wet sediment is transferred to a beaker and stirred for 10 min with diluent. A dilution series of 12 concentrations and 3 controls is prepared from this mixture (Fig. 9). Bacterial reagent is added to the dilutions and incubated at 15°C for 20 min. These dilutions are filtered and the filtrate transferred to the Microtox® analyzer. After 10 min in the analyzer, bioluminescence is recorded (Table 4). Statistics are performed on the data to calculate an EC50 on a wet weight basis (the concentration of test sediment at which light output by a population of the luminescent bacterium was reduced by 50% when compared to the untreated control population).

Table 4. Raw Data used in Fig. 10 Plots. "It" is the Light Reading of a Filtrate of a Particular Concentration of the Test Material. The Microtox Omni Software Automatically Selects the Data to be Used in the Calculation.

Test Treatment	Concentration (mg/L)	It	Gamma	% Effect
Control	0.000	103.37		
Control	0.000	100.32		
Control	0.000	94.86		
1	96.19	89.50	0.1119#	10.07
2	192.4	81.70	0.2181#	17.90
3	384.8	65.78	0.5129#	33.90
4	769.5	42.99	1.315#	56.80
5	1539	15.28	5.513#	84.65
6	3078	1.56	62.79#	98.43
7	6156	0.68	145.3*	99.32
8	12310	0.32	310.0*	99.68
9	24630	0.26	381.8*	99.74
10	49250	0.09	1105*	99.91
11	98500	0.08	1243*	99.92
12	197000	0.07	1421*	99.93

Notes: # — used in calculation; * — invalid data.

For each test concentration, *gamma* (Γ) is calculated as:

$$\Gamma = (I_c/I_t) - 1 \tag{3}$$

where I_c = the average light reading of filtrates of the control solutions and I_t = the light reading of a filtrate of a particular concentration of the test material.[56] The Gamma values for each test filtrate that fall within the range of $0.02 <$ gamma $<$ 200 are plotted. The IC50 is the concentration that corresponds to a gamma of 1.0. If the Microtox® Omni software is not available, data entered for linear regression should be checked against the observed readings to guard against errors in entry and anomalous estimates of IC50. Plots of gamma and the percentage effect vs. concentration generated by the Microtox® Omni software are shown in Fig. 10 from a solid-phase test on a field-collected sediment sample, and the raw data from the test is provided in Table 4.

A linear regression of logC (concentration, on the ordinate) vs. log Γ (on the abscissa) is computed according to the following equation[56]:

$$\log \Gamma = b(\log C) + \log(a). \tag{4}$$

In the above equation, "b" is the slope and "log(a)" is the intercept of the regression line with the ordinate (y-axis) at $\log \Gamma = 0$, corresponding to $\Gamma = 1$. Therefore, "a" obtained as $10^{\log(a)}$ is the IC50.

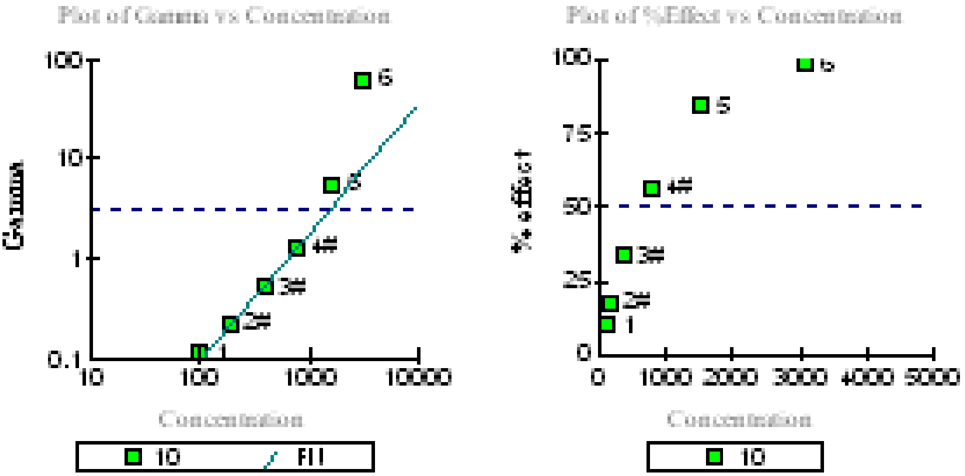


Figure 10. Plots of Gamma Versus Concentration and Percentage Effect Versus Concentration, as Plotted by the Microtox Omni Software. These Plots are from the Raw Data Provided in Table 4, which was also Generated using the Microtox Omni Software using Data from a Test on a Field-Collected Sediment.

The IC50 and its associated values for the 95% confidence limits must be converted to and expressed as mg/kg on a dry-weight basis. The IC50 (as well as the upper and lower values of the confidence limits) of the wet sediment is multiplied by the average of the ratios of the dry-to-wet subsample weights[56]:

$$\mathrm{IC50} = \mathrm{IC50}_w \times [(\mathrm{S1}_d/\mathrm{S1}_w) + (\mathrm{S2}_d/\mathrm{S2}_w) + (\mathrm{S3}_d/\mathrm{S3}_w)]/3, \tag{5}$$

where $\mathrm{IC50}_w$ is the calculated IC50 (or its 95% confidence limits) of the wet sediment sample, $\mathrm{S1}_d$ through $\mathrm{S3}_d$ are the dry weights of the sediment subsamples, and $\mathrm{S1}_w$ through $\mathrm{S3}_w$ are the corresponding wet weights. These calculations can be expedited by entering the weights and IC50 values into a spreadsheet and using the necessary formulae.

A reference toxicant test using a National Research Council of Canada Certified Reference Material (HS-5) is performed in the same manner as the sediments. The IC50 for this analysis is entered into the Shewhart Control Charts to ensure that the test is within standard operating limits, and that the population of bacteria used in the test was of normal sensitivity.

4.2.1.3. Echinoid fertilization test on sediment pore water

The Echinoid fertilization test on sediment pore water has been widely used because of its sensitivity to contaminants. This and other tests on sediment pore water have been thoroughly discussed in Carr and Nipper.[57] The test is summarized in Table 5.

Table 5. Summary of Echinoid Fertilization Test Procedure.

Test organism	Marine echinoids: *Lytechinus pictus*, *Strongylocentrotus purpuratus*, *Arbacia punctulata*, *Dendraster excentricus*, *Strongylocentrotus droebachiensis*
Type of test	20-min (10-min sperm exposure, 10-min sperm plus egg exposure) static test on sediment porewater
Test vessels	10-mL glass scintillation jars
Volume of test solution	10-mL porewater
# Test animals	2000 eggs per test container, 100 scored for fertilization
Lighting	Continuous light
Test temperature	Usually 20 ± 1.0°C
Experimental configuration	Three or more replicates per concentration of porewater
Endpoints	Average % eggs fertilized ± SD IC25 (concentration causing 25% inhibition of fertilization compared to the control)
Measurements	T, DO, pH, salinity, and ammonia in porewater
Reference toxicants	$CuSO_4$ in a water only test

The test species to be used for a particular study is selected based on the availability of gravid animals at the time of the study.[39] Test animals can either be purchased from commercial suppliers, or collected locally, once their identification has been confirmed by a qualified taxonomist. Some species can be cultured in the laboratory for a year round supply of gravid test organisms (*Lytechinus pictus* and *Arbacia punctulata*).

The test is conducted according to Environment Canada guidelines.[39] A portion of each of the sediments is centrifuged at 10,000 g for 15 min at 4°C. The supernatant liquid is transferred to a new centrifuge tube and centrifuged for an additional 15 min at 10,000 g at 4°C to remove remaining suspended sediment particles. Porewater is measured for temperature, pH, salinity, and dissolved oxygen. If dissolved oxygen is low (<40% saturation), the samples are aerated for a maximum of 20 min or until 40% saturation was obtained. Three or more replicates of the dilutions of the porewater and of a control are prepared for each sample. The control/dilution water is laboratory seawater supply filtered through a 0.45-μm filter prior to use, and stored for no more than 24 h.

Sea urchins are injected with 0.5 mL of 0.5 M potassium chloride solution to induce spawning. Eggs produced from all females are pooled, and the concentration is adjusted to 2000 eggs/mL. Sperm are pooled from all males (Fig. 11), and then stored in a vial on ice. A suitable "sperm-to-egg ratio" is chosen to produce approximately 90% fertilization in the controls. Sperm are activated with seawater immediately prior to the start of the test.

Using a test volume of 10 mL at a temperature of 20 ± 1°C, sperm are exposed to the test solutions for 10 min, followed by an additional 10-min exposure of the sperm

Figure 11. "Wet" Spawning of Female (Left) and Male (Right) Sea Urchins.

and eggs (longer exposures for *Arbacia punctulata*). The test is then terminated using 1 mL of 0.5% glutaraldehyde per replicate. One hundred eggs are examined from each replicate at 100 times magnification to check for the presence of a fertilization membrane. Average percentage fertilization and standard deviation for the replicates of each test sediment, reference sediment, and control are calculated and compared for statistically significant differences.[43] The IC25 and IC50 are calculated using the CETIS statistical software.[52] A blank sample containing eggs but no sperm (four replicates) is also analyzed to ensure no false fertilization occurred.

A reference toxicant test is performed simultaneously with the porewater toxicity tests using copper sulfate. The IC50 is calculated using log dose transformed data. The value is entered into the Shewhart control chart to ensure that the test was within standard operating limits, and that the test gametes were of normal sensitivity to the reference toxicant.

4.2.1.4. Growth and survival test with marine polychaetes

Environment Canada developed the growth and survival test with marine Spionid polychaetes because there was a need for a chronic sublethal sediment toxicity test for disposal site monitoring. The test is summarized in Table 6. The polychaetes (*Polydora cornuta*) are cultured in-house from worms originally collected from

Table 6. Summary of the Growth and Survival Test with Marine Polychaetes.

Test organism	Spionid polychaete *Polydora cornuta* (juveniles 0.06–0.5 mg)
Type of test	14-day static test on whole sediment; 75% of the overlying water was replaced with clean seawater on Day 7; fed three times per week
Test vessels	300-mL tall form glass beakers
Volume of test solution	50 mL sediment, 200 mL overlying water
# Test animals	Five per test container
Lighting	16 h light: 8 h dark
Test temperature	23 ± 1.0°C average, 23 ± 3.0°C instantaneous
Experimental configuration	Five replicates per test sediment
Endpoints	Average % survival ± SD; average dry weight per worm ± SD; average biomass
Measurements	T, DO, pH, salinity, and ammonia in overlying water
Reference toxicants	$CdCl_2$ in a water only test

Figure 12. Polychaete Worm *Polydora Cornuta.*

Conrad's Beach, Nova Scotia, Canada. This is to ensure a supply of suitably sized juveniles is available whenever studies are to be carried out (Fig. 12).

The test is conducted according to Environment Canada (2001).[42] Sediment from where the polychaetes were originally collected is used as control sediment. The test is conducted at 23 ± 1°C under a 16-h light: 8-h dark photoperiod. Five replicated

300-mL tall form glass beakers are filled with 50 mL of sediment (control sediment, reference sediment, and/or test sediment) and 200 mL of clean and laboratory supplied filtered seawater is added using a flow diffuser to minimize disturbance of the test sediment. The test vessels are aerated overnight prior to the introduction of test organisms.

On Day 0 (i.e. test initiation), five randomly selected polychaetes (0.06–0.5 mg dry weight) are added to each test chamber. For the duration of the test, the jars are aerated with filtered and oil-free compressed air, at a steady rate of approximately 2–3 bubbles per second. Water quality parameters (pH, dissolved oxygen, temperature, and salinity) are measured three times per week throughout the 14-day test period. Approximately 75% of the overlying water is replaced with clean seawater on Day 7 of each 14-day test. Every Monday, Wednesday, and Friday, each group of test organisms is fed 2 mg (dry weight) per worm of food (50:50 by weight blend of ground green alga *Enteromorpha* spp.: Tetramarin® commercial fish flake).

On Day 14, a sample of overlying water is taken and analyzed for ammonia. The sediment in each test chamber is sieved (500-μm sieve) and the worms removed from their tubes using gentle pressure from a stiff and fine artist's paintbrush. The number of surviving, dead, and missing polychaetes is recorded. Surviving polychaetes are rinsed in deionized-distilled water before placement in pre-weighed aluminum pans for final dry weight determinations (after drying at 60°C for $\geq$ 24 h).

The means and standard deviations are calculated for percentage survival and growth of replicate groups of spionid worms exposed to each test, reference, or control sediment and compared for statistically significant differences.[43]

The LC50 value (and associated 95% confidence limits) for the positive control reference toxicity test (using cadmium chloride) is determined using the Environment Canada computer program based on the methods of Stephan 1977.[51] Results obtained for the reference toxicity test using this program are entered on a Shewhart control chart to ensure that the test was within standard operating limits, and that the test organisms were of normal sensitivity to the reference toxicant.

4.2.1.5. Bedded sediment bioaccumulation test

Many studies have concluded that only a fraction of the total concentration of organic substances (i.e. the bioavailable fraction) is available for uptake and has the potential to cause toxicity. The most direct means to determine biologically relevant sediment concentrations is by measuring the contaminant uptake by receptor organisms, either in resident populations in the field, or by means of laboratory bioaccumulation tests. These studies allow for the calculation of biota-sediment concentration ratios (e.g. BSAFs), and provide insight into the bioavailability of contaminants in a given sediment. A more recent approach to investigate contaminant bioavailability

Table 7. Summary of the Bedded Sediment Bioaccumulation Test Procedure.

Test organism	Usually marine clam *Macoma balthica* (other species acceptable)
Type of test	28-day static test on whole sediment; three times a week approximately 75% of the overlying seawater renewed
Test vessels	4-L containers
Volume of test solution	1–2 L sediment, 2–3 L overlying water
# Test animals	20–60 per test container (depends on contaminant of concern)
Lighting	16 h light: 8 h dark
Test temperature	Usually 15 ± 2.0°C
Experimental configuration	Three or more replicates per test sediment
Endpoints	Average % survival ± SD; uptake of contaminants into clam tissues
Measurements	T, DO, pH, salinity, and ammonia in overlying water
Reference toxicants	None identified

is through the use of passive samplers, often referred to as "equilibrium sampling devices." These techniques involve bringing small volumes of a lipid-surrogate to equilibrium with a contaminated environment, and using the resulting sampler concentrations to approximate uptake by biota. Equilibrium sampling devices have been applied in various environmental media including sediment porewater,[58,59] and sediments.[60] Application of these techniques may provide a simple and cost-efficient means to approximate the bioavailability of hydrophobic contaminants in sediments. The test described in this chapter is the bedded sediment bioaccumulation test using the clam, *Macoma balthica*,[40] The method is summarized in Table 7, and the test organism is shown in Fig. 13.

M. balthica has not been cultured in the laboratory to date, and it would be impractical to do so because of its long life span. It must be collected from the field or purchased from a supplier. The cultured organisms are acclimated to and maintained at the test conditions (15 ± 2°C and 30 ± 2 ppt salinity) until used for testing.

Each container of test sediment is homogenized and 1–2 L portions are added to each of three or more plastic pails. The pails are then filled with 2–3 L of clean seawater (salinity was 30 ± 2 ppt), then aerated overnight with oil-free compressed air at a rate of approximately 150 mL/min. Tests are conducted according to USEPA (1993).[40]

The following day clams are removed from their holding sediment by sieving the contents through a 2000-μm sieve. Animals are double-counted and sufficient animals are added to each of the test vessels to supply tissue for chemical analysis. Testing is performed with a 16:8 h light:dark photoperiod with lighting provided by overhead fluorescent fixtures at an intensity of 100–500 lux. Testing is performed at

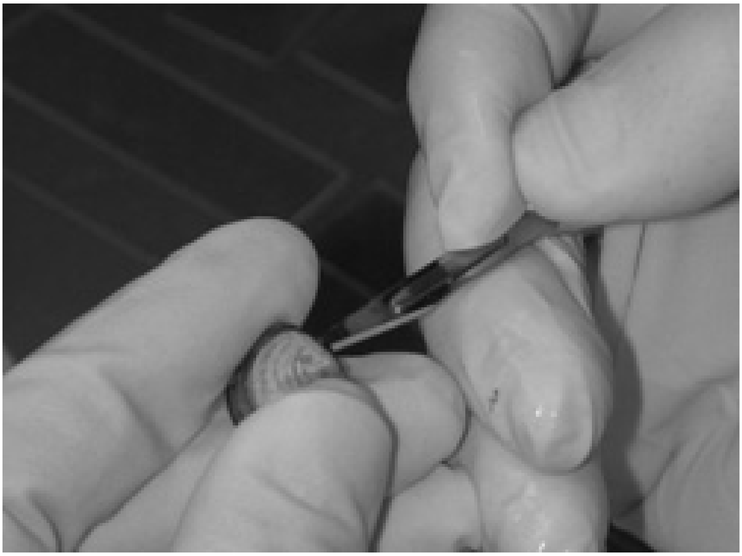


Figure 13. The Baltic Clam, *Macoma Balthica.* Removal of Tissue for Bioaccumulation Study.

15 ± 2°C. The tests are checked daily for observations, aeration, and temperature. Three times a week a replicate of each sample is measured for temperature, pH, salinity, and dissolved oxygen, and less frequently for ammonia. Three times a week approximately 75% of the overlying seawater in the test chambers is renewed.

At the start of the test, three sets of animals are removed and tissue is collected and preserved by freezing for chemical analysis of contaminants of concern. The lengths and weights are measured on one set of these animals. After 28 days, the contents of each container are sieved through a 2000-μm sieve. The number of surviving clams is recorded. The mean survival and standard deviation of each treatment are calculated. The clams are then transferred to collection site sediment for 1 day to depurate gut content. On day 29, the animals are sieved from the collection site sediment and tissue removed (Fig. 13), rinsed in distilled water, and preserved for chemical analysis.

Sediment bioaccumulation factors (BAF) and organic carbon and lipid normalized biota-to-sediment accumulation factors (BSAFs) are calculated using the following equations:

$$\mathrm{BAF} = C_{\mathrm{tissue}}/C_{\mathrm{sed}} \tag{6}$$

$$\mathrm{BSAF} = (C_{\mathrm{tissue}}/C_{\mathrm{sed}})^{*}(f_{\mathrm{toc}}/f_{\mathrm{lipid}}), \tag{7}$$

where C_{tissue} is the concentration in the tissue (in μg/kg wet weight), C_{sed} is the measured sediment concentration (in μg/kg dry weight), f_{toc} is the fraction of TOC in the test sediment, and f_{lipid} is the fraction of lipid in *Macoma* tissue. Equation (6) calculates the ratio between tissue and sediment concentrations, while Eq. (7) expresses this ratio in terms of lipid and organic carbon concentrations. The lipid and organic carbon normalized BSAF is commonly used when studying hydrophobic chemicals, since these chemicals accumulate primarily in organic materials.

4.2.2. *Benthic Infauna Surveys*

In addition to the biological tests, a survey of benthic infauna is probably the most commonly and widely used assessment tool for assessing dredged material disposal impacts at nearshore disposal sites.[61–65] Benthic biota, especially the infauna that are living in the sediment, are sessile and community measures are considered useful *in-situ* measurement parameters for the detection of disturbance caused by physical and/or chemical changes in the marine environment.

Sediment samples for benthic community studies are normally collected by grab samplers, such as the Ekman, Ponar, Van Veen grabs, and the Box corers.[32–34] The selection of a suitable sampler depends mostly on the physical characteristic of the sea bottom. An ideal sampler should consistently collect a similar volume of sediment and disturb the surface layer of the sample as little as possible. To obtain the biota samples, grab samples are sieved through a 0.5- or 1.0-mm sieve. The biota captured by the sieve are then fixed in 10% formalin and later transferred to 70% isopropanol or 70% ethanol for preservation. The preserved samples are later sorted under a stereomicroscope, identified, enumerated, and weighed. Organisms are identified to an appropriate taxonomic level, typically to species using conventional literature for the groups involved. For later verification, a set of representative species are kept as reference. Sometimes, for difficult groups or species, the samples are cross checked against reference species kept in local museum or research centers. Additional QC measures include resorting of 10–20% of already processed samples (to determine sorting efficiency) and cross checking and independent verification of data records.

Benthic biota data collected from a disposal site are used to calculate species abundance, number of species, diversity indices, and biomass and generate various statistical calculations for community disturbance measurement.[66–68]

Typically used community measuring tools include the following:

- *Shannon-Wiener Index* (H') — The index is defined as:

$$H' = -\Sigma(p_i \log_{10} p_i), \tag{8}$$

  where p is the proportion of the number of individuals of a species i to the total number of individuals in the sample.

  This index is widely used in ecology and represents both the number of species and distribution among individuals, with higher numbers of species generally resulting in increased diversity and high values of single species resulting in low diversity.

- *Pielou's Evenness Index* (J') — The index is defined as:

  $$J' = H'/log_{10}S, \tag{9}$$

  where S is the total number of species present in the sample.

  This index is commonly used to express equitability of distribution of individuals among species.
- *McIntosh's Index* (M) — The index is defined as:

  $$\frac{N - \surd(\Sigma n_i^2)}{N - \surd N}, \tag{10}$$

  where N is the total number of organisms in the sample and n_i is the abundance of each species.

  This index measures evenness and can be used as indication of whether the species are present in about the same numbers or single species dominates in the sample. The index falls in a range of from zero to one, reaching a maximum if all individuals are present in perfectly equal numbers.
- *Margalef's Index* — The index is defined as:

  $$\frac{S - 1}{lnN}, \tag{11}$$

  where S is the total number of species and N is the total number of organisms in the sample.

  This index measures the number of species per individual (species richness). It is generally higher when more species are present.
- *Simpson's Index* — The index is defined as:

  $$\Sigma p_i^2, \tag{12}$$

  where p_i is the proportion of the number of individuals of a given species to the total number of individuals in the sample ($p_i = n_i/N$.)

  This index measures the dominance of species. It is higher when a few species make up a large proportion of the individuals in a sample.

In addition to the indices described above, the most commonly used statistical methods for benthic community studies include hierarchical clustering, ordination, dominance curves, species aggregation, and other univariate and multivariate measures of community stress. For the ease of performing these community measures and data interpretation, Clarke and Warwick[68] of the Plymouth Marine Laboratory in the United Kingdom have developed a desktop computer software, the PRIMER, to assist environmental scientists and managers to understand their data and to explain their data in a nontechnical way to others and have confidence that they are using the right methods for the right problem.

Simple statistics such as percentage polychaetes, percentage mollusks, percentage crustaceans, and percentage echinoderms have also been used as a general indicator of benthic community diversity and health.

An effective biological monitoring program requires the combination of the use of proper community measures and statistically based sampling designs. In addition, it is important to recognize that some environmental factors, such as temperature, salinity, dissolved oxygen, nutrients, water depth, sediment organic content, particle size, and season, can confound the interpretation of the benthic community results.[69] Special care should be taken to eliminate these confounding factors when designing the monitoring program and selecting reference sites. Other monitoring tools that can be quite effective for biological survey include the use of bottom photographs, video, and scuba diving. This supplementary information is important for data interpretation of benthic community studies.

Kostylev *et al.*[22] interpreted geological and habitat data from bottom photographs and combined these information with geological data collected by grab samples and by acoustic survey to classified bottom type and biological communities at dredged material disposal sites. For over 20 years, the US Army Corps of Engineers (USACE) adopted a very effective method, the REMOTS@ system, in their DAMOS for routine dredged material disposal site monitoring. The system is designed to obtain undisturbed, vertical, and cross-sectional photographs (*in-situ* profiles) of the upper 15–20 cm of the seafloor. Images obtained by REMOTS@ yield a suite of standard parameters that can be used to map the distribution of thin (<20 cm) dredged material layers, delineated benthic disturbance gradients, and monitor the process of benthic recolonization following the disposal of dredge material.[70–72]

5. Case Studies

The following case studies are selected to demonstrate how the different methods and technologies described in the previous sections are used for dredged material disposal site monitoring. Each of these studies represents a unique situation where flexibility is needed to select the most suitable monitoring tools to achieve cost-effective results.

5.1. *Amherst Cove Dredged Material Disposal Site, Prince Edward Island (PE), Canada*

The Amherst Cove disposal site (46°09.10′N; 63°46.00′W) is located about 2–3 km from terminus of the Confederation Bridge, and near the old ferry terminal in

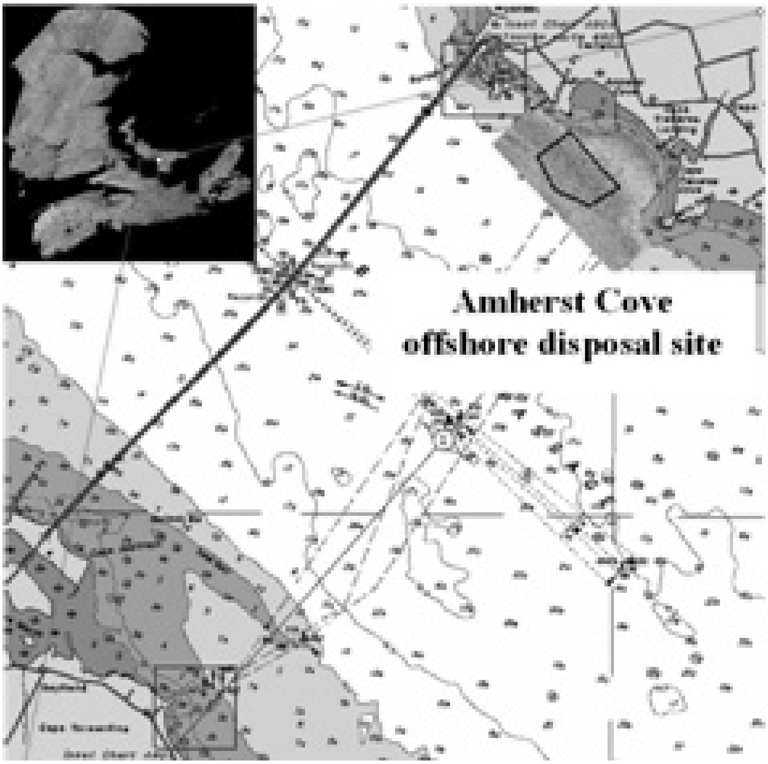


Figure 14. Color-Shaded-Relief Image Created from Multibeam Bathymetry Data Collected in the Amherst Cove Disposal Site, Prince Edward Island, Canada. The Location of the Designated Disposal Site is Shown by the Black Rectangle. The Location of the Survey Area is shown on the Inset Map of Eastern Canada.

Borden, on PE in Atlantic Canada (Fig. 14). The site is located in shallow water, depth ranges from 3 to 11 m, and was designated by Environment Canada in 1993 for the disposal of 473,000 cubic meters of sediment from 59 pier base locations and a jetty required for the construction of the nearby Confederation Bridge. The dredged material consisted of sand and silty sand with gravel, boulders, and pieces of bedrock.

The disposal site was selected after consultation with the local commercial fishing community and was thought to be in an area of poor lobster habitat that might benefit from the deposit of coarser materials. For mitigation purpose, a dredged material disposal plan was developed for the use of this site with two objectives: (1) to create disposal areas that are stable in the dynamic physical marine environment and (2) to enhance crustacean (lobster and rock crab) habitat. Creating stable disposal areas was the primary goal of the disposal plan while depositing the dredged materials in a pattern expected to enhance crustacean habitat was the secondary goal. The specific objectives of the habitat enhancement methods at the site were to:

- introduce coarse material (sandstone, mudstone, cobble, or gravel), that is not abundant in the disposal area, to form substrate that can be colonized by crustaceans;
- place coarse material in sections of the disposal areas adjacent to existing crustacean habitat to facilitate colonization by emigration; and
- place coarse material in a pattern, so that the material efficiently covers the disposal area section of potential habitat enhancement.

Between 1993 and 1996, bathymetric data were obtained at the site before and after the disposal as required by the dredged material disposal plan. In 2001, Environment Canada decided to monitor the site to confirm that disposal was conducted within the disposal boundary and to evaluate the site for shellfish habitat enhancement. Since the dredged materials were clean sands, gravels, and bedrock, only physical and biological monitoring surveys were performed including multibeam, side scan sonar, bottom photography, seafloor video, and sediment grab samples.

As a joint program between Environment Canada and Natural Resources Canada, sidescan sonar and multibeam bathymetry surveys were performed in October 2001.[73] The surveys from 2001 were compared with earlier multibeam bathymetry data collected in 1995 and confirmed that the majority of the dredged material was placed within the designated disposal site (Figs. 15 and 16). The material was still quite prominent six–eight years after placement on the seafloor, indicating that erosion by the currents in the area has not had a large effect on the features. Sections of the disposal site in water depths greater than about 10 m had been observed to contain sand ribbons and comet marks on the sidescan

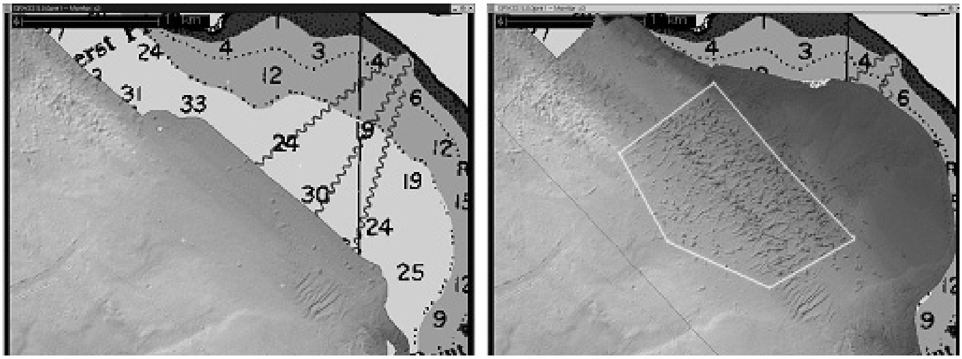


Figure 15. Amherst Cove Disposal site Multibeam Images Collected in (a) 1995 and (b) 2001.

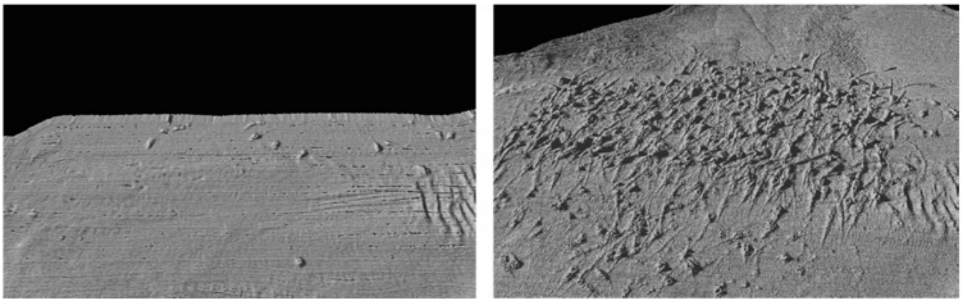


Figure 16. (a) Amherst Cove Disposal site 3D Multibeam Images Collected in (b) 1995 and 2001.

sonar records, indicating the periods of sediment transport (in this case to the southeast). These bedforms were generally formed in sediment starved areas. The sections of the disposal site with water depths less than about 10 m did not show the indications of active sediment transport. Comparison of the location of dredged materials from multibeam bathymetry surveys performed in 1995 and 2001 indicates that the dredged material detected at the disposal site in 1995 is still present.

Seafloor photographs and video transects showed that the spoils were made up of a broad spectrum of grain sizes including large angular boulders and large pieces of rock. The coarser material should remain in place for a considerable time. Redistribution of the coarse material by ice scour during the winter months would be the most likely cause of movement in this location. Seafloor photographs show a marked change in the water clarity, possibly due to seasonal variations, from June 1995 to October 2001. The photographs from October 2001 show an accumulation of fine-grained sediment overlaying much of the dredged material. It is anticipated that much of the fine material will be resuspended and transported out of the area during winter storms.

Both the video survey and bottom photographs were analyzed by fisheries biologists specializing in crustacean habitat studies. The analysis showed a slight improvement in lobster habitat by the addition of the dredge spoils, and a significant improvement in rock crab habitat. It is likely that the presence of the fine-grained material seen in the October 2001 survey had reduced the suitability of the habitat for lobster at that time of the year, while rock crabs thrive in these conditions. During video transects through the area, rock crabs were 10 times more common than lobster. Another video survey is recommended for earlier in the year to study the colonization by lobster under more favorable circumstances.

This case study demonstrated the use of physical monitoring tools to test two hypotheses: (1) dredged materials were disposed within the boundary of the disposal site and that the site is a stable site and (2) the disposal has improved the poor crustacean habitat at the disposal site. The data collected from this study can be used by disposal at sea managers to make appropriate management decision in selecting suitable disposal sites when they encounter similar dredging projects in the future.

5.2. *Canso Common User Dredged Material Disposal Site, Strait of Canso, Nova Scotia, Canada*

The Strait of Canso separates Cape Breton Island from mainland Nova Scotia (Fig. 17).[20] Prior to the completion of the Canso Causeway in 1955, the Strait had strong currents with a strong net water flow southward. High current velocities (2.67 m/s (9.6 km/h) south-going and 2.1 m/s (7.6 km/h) north-going) within the Strait of Canso resulted in erosion of seafloor sediments and generally maintained a coarse substrate by preventing deposition of fine-grained marine sediments on the seafloor. After completion of the causeway, the Strait of Canso exhibited conditions more closely related to a tidal inlet, or fjord. The quieter current regime is prone to deposition of fine-grained sediments resulting in a change from an erosional environment that saw the development of sandwaves and erosional flutes, to a depositional environment with the accumulation of fine-grained sediments at rates in the range of 1–2 mm per year.

The Canso common user dredged material disposal site (45°35.65′N; 61°22.67′W) is located in a 650-m long and 200-m wide trough with depth of 55–64 m within the Strait of Canso. When the site was chosen for dredged material disposal, it was determined that it has a capacity for holding 500,000–600,000 cubic meters of dredged material. To date, approximately 97,000 cubic meters of dredged materials have been disposed of at the site. Most of these materials came from nearby dredging operations along both sides of the strait. Chemical analyses indicated that some dredged materials contained concentrations of PCBs and Cd above Environment Canada's screening criteria of 0.10 mg/kg and 0.60 mg/kg, respectively.

Between 1985 and 1987, without the help of acoustic surveys, samples were collected at the disposal site and the surrounding areas to assess the levels of contaminants in the sediments and the benthic communities.[74,75] The 1985 survey showed that the concentrations of Cd, Zn, Cu, and Pb were higher in the disposal site sediments than those measured before disposal. However, with the exception of Cu, these differences were not statistically significant. High levels of PCBs, up to 1 mg/kg, were detected in samples collected from the disposal site. Repeat sampling in the 1987 survey showed a wider distribution of PCBs in the strait, not just restricted within the disposal site. The highest concentration of PCBs (2.6 mg/kg) was in the samples collected along the eastern shoreline of the strait in front of a forest products plant.

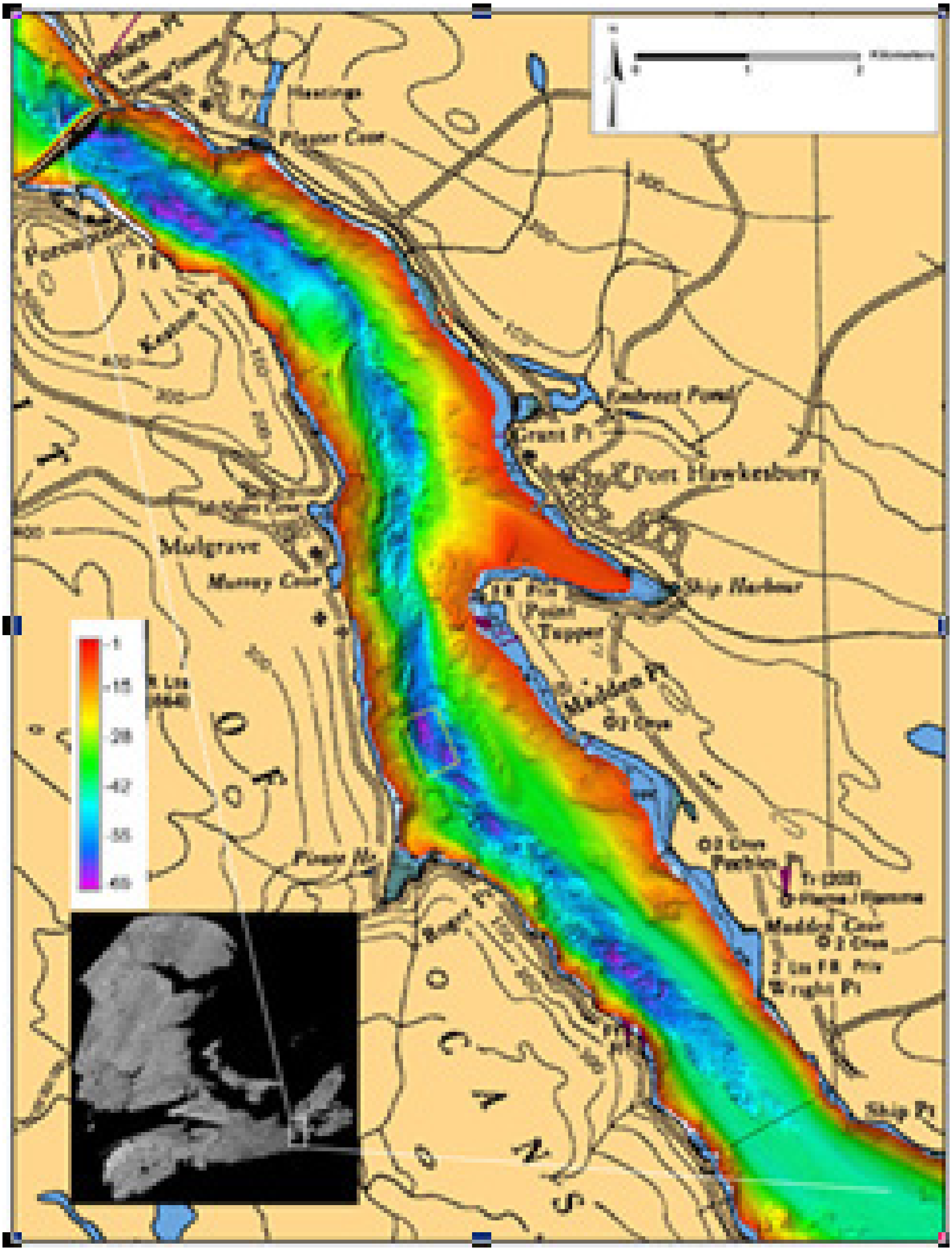


Figure 17. Color-Shaded-Relief Image Created from Multibeam Bathymetry Data Collected in the Strait of Canso in 2004. The Location of the Designated Ocean Disposal Site is shown by the Yellow Rectangle. The Location of the Survey Area is shown on the Inset Map of Eastern Canada. The Color Bar Used for the Bathymetry Data is shown on the Left Side of the Image.

Both the number of species and the abundance of polychaetes increased from 1985 to 1987, indicating that, although the original fauna was partially disturbed at the disposal site, rapid recolonization had taken place. *Prionospio steenstrupi*, a second-stage recolonizer was the dominant species both in 1985 (44%) and 1987 (36%). *Spiophanes kroyeri* also occurred regularly and in high numbers. Other second-stage recolonizers collected in 1987 were members of the polychaete genera, *Lumbrineris*, *Eteone*, *Schistomeringos*, *Pholoe*, and *Nereis*. The increased number of species and individuals, especially the second-stage recolonizers, indicated that the infaunal community at the disposal site had undergone a recolonization process. Log-normal distributions of the infaunal data indicated that the community was still in the recovering process in 1987. The results of cluster analyses that were conducted on the 1985 and 1987 infaunal data showed that there is a distinct group of infauna located at the center and south end of the disposal site where sediment samples are mostly sand and gravel. This related to the area where most dumping took place. The cluster patterns are also consistent with the depth contour of the 1987 sounding survey at the site.

In a recent disposal site survey conducted by a permit applicant, the elevated concentrations of Cd and PCBs were still detected in the disposal site samples. Benthic community data showed that the most dominant species at the disposal site were *P. streenstrupi*, *Goniada maculate*, and *Nephtys neotena.*

In 2004, based on the requirement of the 1998 Environment Canada National Guidelines for Monitoring Dredged and Excavated Material at Ocean Disposal Sites, a monitoring program was carried out by Environment Canada to evaluate the disposal impacts at the site. In following the guidelines, a physical monitoring was first conducted at the site to delineate the disposal areas following by chemical and biological monitoring to assess the chemical and biological impacts.

The same suite of physical monitoring tools used at the Amherst Cove site was used at the Canso site.[20] The data provides an interesting insight into the present and past conditions on the seafloor of the Strait. The accumulation of sediments is presently continuing, with a veneer of fine-grained material covering most of the deeper portions of the Strait. The source of the most recent sediments south of the causeway is mainly local, with material derived from natural (e.g., eroding shorelines, small drainage systems) as well as anthropogenic (e.g., urban and industrial wastes) sources. The Strait is currently influenced by ongoing sea level rise and has risen from an early Holocene lowstand at about -38 m. The mean sea level continues to rise at >0.4 m/century.

Multibeam bathymetric data collected in the area showed large-scale features such as erosional flutes and sandwaves, present in the central portion of the Strait (Fig. 18). The large u-shaped bedforms, known as erosional flutes, are formed through the erosion of muddy sediment surfaces by eddies in a strong current

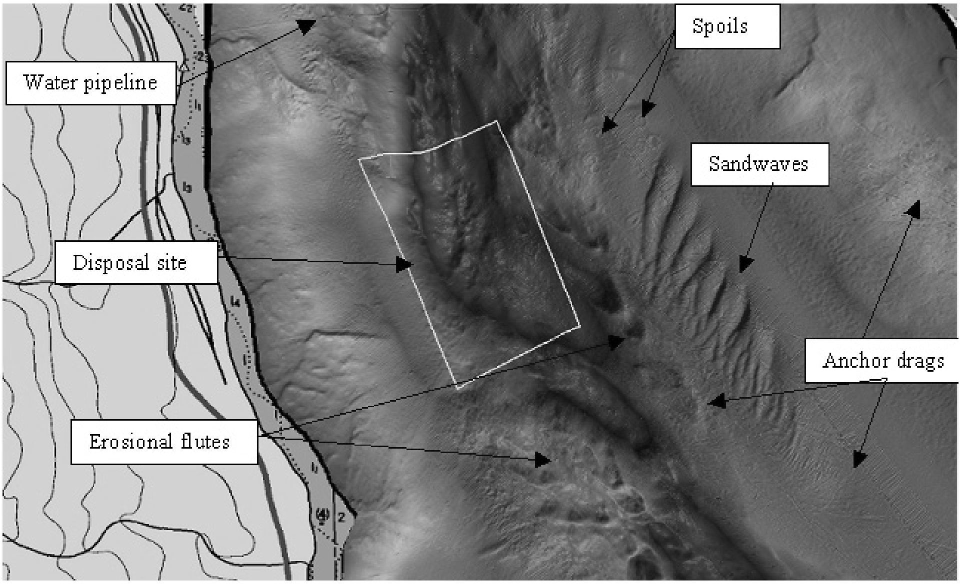


Figure 18. Multibeam Bathymetric Data Collected in the Strait of Canso Showed Large-Scale Features such as Erosional Flutes and Sandwaves, Present in the Central Portion of the Strait.

transporting sediment toward the open end of the flute. Sandwaves are formed by strong currents acting on sandy sediments and are generally asymmetrical, with steep lee (downstream) and gentle stoss (upstream) sides. Both of these features are indicators of strong current flow in the area. However, seafloor samples, video, and images show much of the seafloor in the Strait near the disposal site is covered with a veneer of fine sediments, indicating a quiet current regime at the present time. Based on the sample and photographic evidence, both the flutes and sandwaves probably pre-date the construction of the causeway.

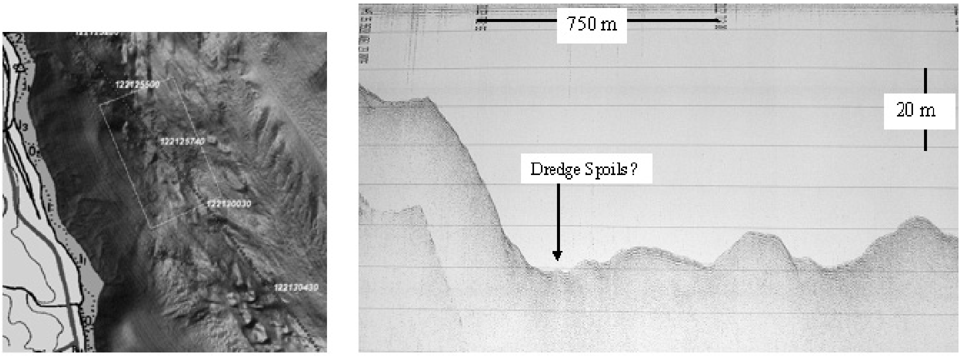
Analysis of the multibeam bathymetry data also shows the presence of numerous small-scale features such as anchor furrow marks and depressions left by the spud cans from jack-up oil rigs and dredge barges. A rough area on the seafloor near a local gypsum plant (not shown in the figure) is characteristic of seafloor that has been dredged and may be the source of about 40,000 m^3 of the material in the disposal site.

Multibeam images show that the dredged materials are characterized by zones with rough surfaces in the trough near the center of the disposal site. Most of the materials are within the disposal site boundary with evidence of the circular patterns associated with dredge spoils detected in several locations just east of the disposal site. The faint circular patterns of materials may be an indication that spoils were disposed at this site prior to the controlled disposal in the Canso Strait. A zone of high acoustic backscatter within the disposal area may be related to the disposal of coarse material at the disposal site.

Other geophysical equipment used during the monitoring survey consisted of a dual-frequency (120 and 330 kHz) sidescan sonar system, and an IKB Seistec® sub-bottom profiler collected on a 75-m grid. Sidescan sonar data from the area were processed and combined into a mosaic to provide an overview of the character of the seafloor sediments. The sidescan sonar data provide additional information on both the distribution of sediments, and fine-scale features such as dredge spoils and anchor furrows, with a higher resolution than that obtained with the multibeam bathymetry data. The two data sets complement each other.

Sub-bottom profiler data provide information on the thickness of surficial sediments and insight into the nature and genesis of sediments below the seafloor. The seafloor at the disposal site is composed of coarse sediments, probably glacial till, overlying bedrock. An accumulation of about 1 m of recent sediments in this area is probably related to the recent disposal of dredge spoils at the site as shown in Fig. 19. Adjacent to the disposal site, thick deposits of glaciomarine or glaciolacustrine sediments have been eroded to form flutes. Several episodes of erosion and deposition are evident. A protective berm over a natural gas pipeline is quite evident. The stratified glaciomarine/glaciolacustrine sediments generally terminate at a regional unconformity at — 38 m, indicating a lowered sea level to this depth.

Sediment samples were collected throughout the area, to assist with interpretation of the geophysical and multibeam bathymetry data and to study sediment composition and the biological communities identified on photographyic and video images. Historical sample data were recovered from the Geological Survey of Canada on-line database. The samples provide information on the nature of sediments throughout



(a) (b)

Figure 19. (a) Location and (b) Seistec® Sub-Bottom Profiler Data through the Disposal Area. The Seistec® Data is shown from Day 122 12:53 to 13:05. An Accumulation of about 1 m of Recent Sediments in this Area is Probably Related to the Recent Disposal of Dredge Spoils at the Site.

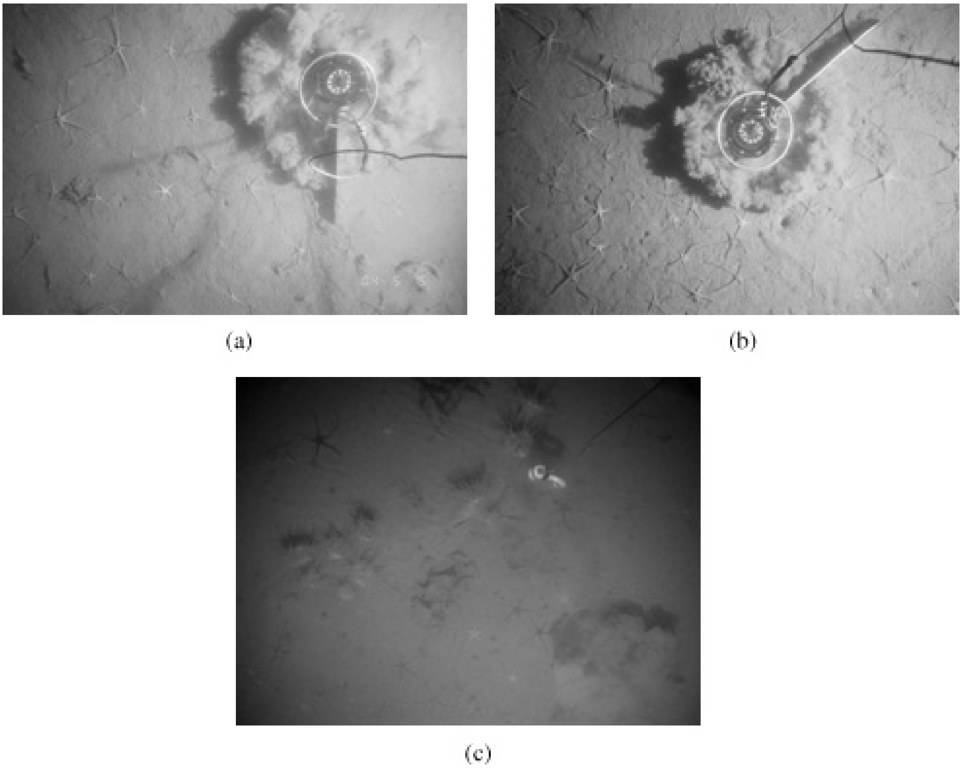

(a) (b) (c)

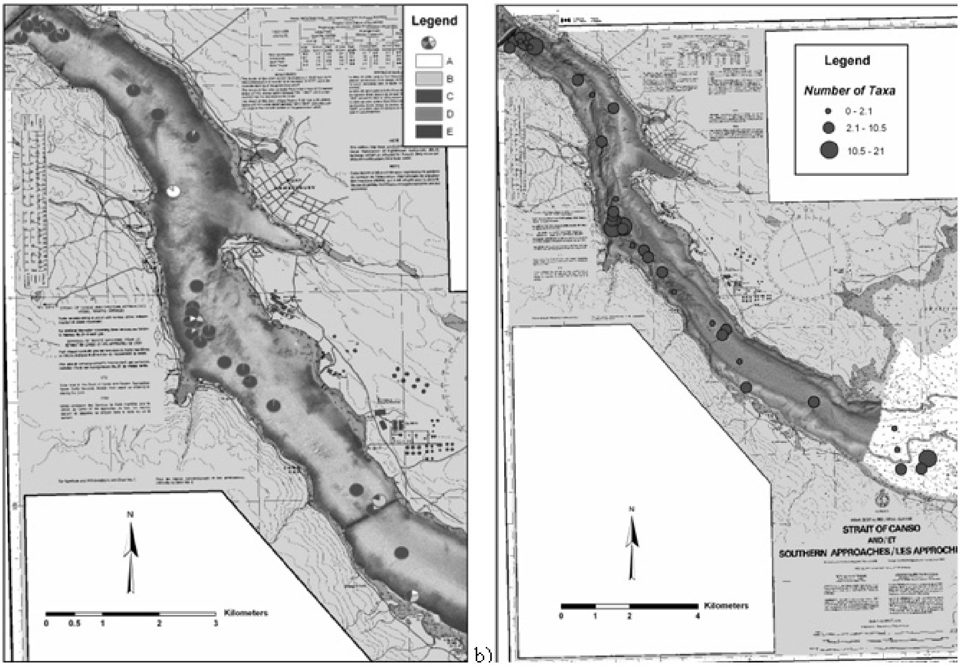
Figure 20. Seafloor Photographs from (a) within and (b) Near the Disposal Site. Sea Anemone (c) were Present in Many Areas, Indicating the Presence of Hard Substrate Under the Fine Sediments.

the Strait with emphasis on the area around the disposal site. All samples were analyzed for grain size; many were also analyzed for carbon content and metal concentrations. Most of the recent samples from within the Strait contained a veneer of light brown and fine-grained sediment overlying coarser material. Samples from within the disposal site generally contain a small percentage of sand with some gravel present.

Images of the seafloor (Fig. 20) were obtained with a variety of systems including the Fisheries and Oceans TOWCAM towed camera system[76]; a 35-mm film camera; and an underwater video and digital still camera system. Physical habitat characteristics and the type and relative abundance of benthic fauna were interpreted from the seafloor photographs. Features such as counts of animals and burrows were classified. Physical habitat was described in terms of relative abundance of boulders, cobbles, pebbles, granules, sand, and silt and perceived topographic complexity. Other habitat descriptors were surface complexity and water turbidity, as well as abundance of shell hash, large and small burrows, and siphons of infaunal invertebrates. All visible species of megabenthos were identified to the highest possible taxonomic resolution. Average abundance ranks and habitat characteristics were calculated for each seafloor photograph.

Cluster analysis and nonmetric multidimensional scaling plots were generated from the interpretation of benthic organisms seen on the photographic data. Five distinct clusters were distinguished based on the co-occurrence of benthic organisms on seabed photographs. Two community types were determined to be most common and widespread in the study area — a soft-bottom community (assemblage C) dominated by the brittle star *Ophiura sarsi* and a hard bottom community (assemblage A) mainly comprising brown algae and horse mussels (*Modiolus modiolus*).

On the photographic images, the richness of benthic epifauna was found to vary with the substrate type and distance from the Canso Causeway. Assemblage A is common on hard substrates such as gravel; assemblage C is common on soft and muddy sand. Other assemblages do not exhibit a distinct pattern and are less represented through the area. Assemblage A was observed to increase with increased gravel content accompanied with a corresponding decrease in assemblage C. Figure 21(a) shows the distribution of the clusters overlain on the acoustic



(a) (b)

Figure 21. (a) Relative Frequency Distribution of Assemblage Types in the Strait of Canso. Assemblage C is Found in the Areas of Low Acoustic Backscatter (Lighter Colors). Assemblage A is Concentrated in the Areas with Higher Backscatter (Darker Colors). (b) Number of Benthic Taxa Observed on the Seabed in the Study Area Overlain on Depth.

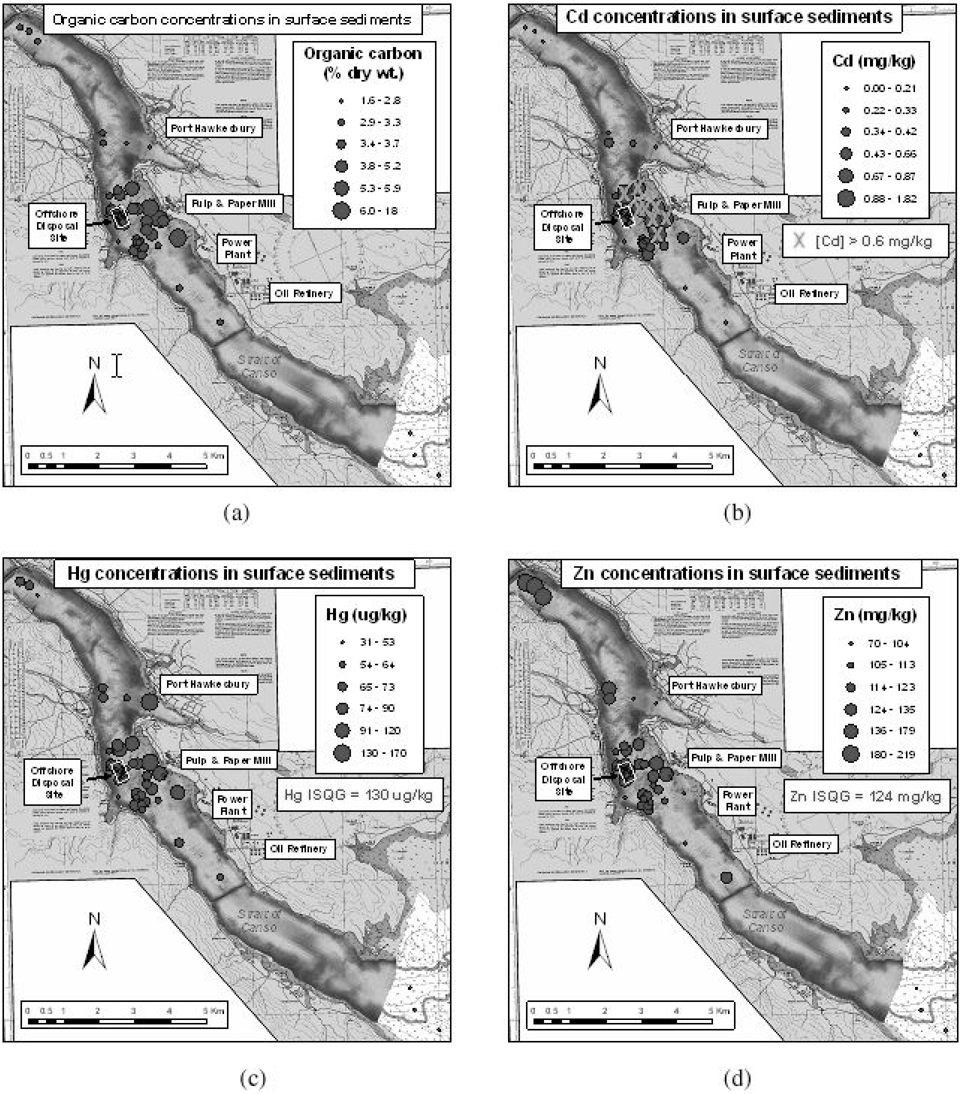
backscatter intensity. Assemblage C, which contains the *Ophiura*, is the most common assemblage and is found throughout the Strait in much of the deeper water, with a soft substrate (indicated by lighter colors on the acoustic backscatter data). Near the disposal site, assemblage C dominates, with some occurrence of species from other groups. Hydrozoans and bryozoans are commonly found on gravel particles here, and may be related to gravel occurrences due to disposal activities. Near the disposal site, there were fewer indications of bioturbation (such as infaunal burrows) than in other soft-bottom sites in the Strait, suggesting possible effects by disposal activities. Assemblage A is seen to occur near areas with high backscatter (dark gray color on Fig. 21(a)) indicating the presence of coarser sediments such as gravel.

Of the images analyzed, biodiversity was the highest in the areas immediately outside of the Strait where a maximum of 21 macrobenthic species were observed per station (Fig. 21(b)). Biodiversity decreased within the Strait. The amount of seabed covered by bryozoans and hydroids, and other epibenthic species, also increased with proximity to open water. Within the Strait and approaches, epifauna was directly related to sediment texture and became more diverse with the increase in seabed surface cover by cobbles and boulders.

The number of taxa observed at each station in the strait increases with water depth, and with the fraction of sediment cover by muddy sand where the number and variability of taxa is the highest. Generally, the number of taxa in the study area is highest on muddy sand, relating to higher diversity of *Ophiura* assemblage. The disposal site did not show a distinct difference from other stations in taxonomic diversity.

Proportional dot size plots have been prepared for several heavy metals and TOC to display the spatial trends in metal(loid) distributions in surface sediments from the Strait of Canso (Fig. 22). In each of these plots, the data have been separated into six percentile classes (0–25th, 25–50th, 50–75th, 75–90th, 90–95th, and 95–100th) to help identify the sources and distribution of each element.

Figure 22 shows data for TOC, Cd, Hg, and Zn. In all cases, sediments near the forest products plant outlet contain much higher concentrations of these elements than most other areas of Canso Strait. Eight of these sites (highlighted with a green "X") have Cd concentrations greater than the current ocean disposal guideline of 0.6 mg/kg. However, the samples taken from within the disposal site generally have relatively low concentrations of most metal(loids), which may reflect the coarse-grained nature of the dredge spoils, and deposition of natural marine sediments dredged from various areas. As shown in Fig. 22, there are also other industrial sources along the eastern shore of Canso Strait that may contribute metal(loid)s to the marine environment; however, the sampling density in this study was not sufficient to characterize the environmental footprint of these sources.



(a) (b)

(c) (d)

Figure 22. (a) Organic Carbon, (b) Cadmium, (c) Mercury, and (d) Zinc Concentrations in Surface Sediments (0–5 cm) Collected from the Strait of Canso in 2004.

Following the physical monitoring, sediment samples were taken from the disposal site, a nearby reference site, nearfield site, and outside areas to conduct chemical analyses and toxicity tests. The nearfield and outside samples were collected to check the surrounding areas outside of the disposal site. Sediment samples taken from the disposal site have low concentrations of heavy metals and PCBs with one sample showing 3.13 mg/kg of PAHs, which is slightly over the Canadian Disposal at Sea screening value of 2.5 mg/kg (Fig. 23). However, the mean PAHs in the

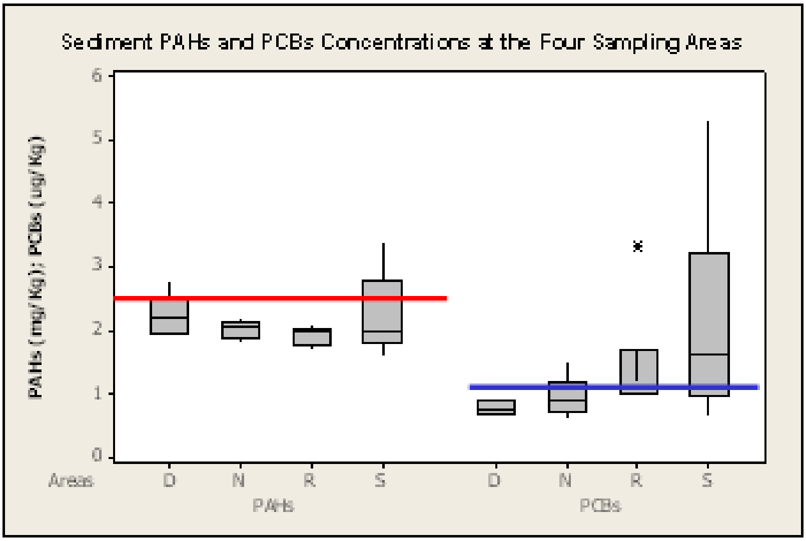


Figure 23. Sediment PAH and PCB Concentrations at the Four Sampling Areas. The Red and Blue Lines are the PAH and PCB Environment Canada Disposal at Sea Limits (PAH = 2.5 mg/kg; PCB = 1 mg/kg); (D = Disposal Site; N = Nearfield Site; R = Reference Site; S = Outside Areas).

disposal site samples are not significantly different (ANOVA test, $P = 0.471$) from those detected in samples from the other studied sites. A suite of sediment bioassays recommended by the 1998 Environment Canada monitoring guidelines[18] was performed on the samples. The suite includes an amphipod (*E. estuarius*), Microtox® Solid-phase, Sea Urchin fertilization test, and polychaete survival and growth test. All test results indicate that the disposal site sediments are nontoxic to the test organisms except in two disposal site samples where survival rates of polychaetes are quite low (20–40% survivals) and all but one of the nearfield site sample in the sea urchin fertilization test. However, because of the existence of high levels of ammonia and sulfide in some of the test samples, it is suspected that the cause of the toxic effect may be triggered by the ammonia and sulfide since none of the endpoints of the tests are statistically correlated to the sediment contaminants. In addition, the mean values for the six stations taken in each of the three main study areas (disposal site, reference site, and nearfield site) are not significantly different and revealed no significant toxic effects at the disposal site compared to the other study areas (Table 8).

Because toxicity was detected in some sediment samples and one of the disposal site samples contains high PAHs, additional analyses were conducted on the benthic biota data collected from the disposal site and all the other studied sites by following the tiered monitoring approach. Results of the benthic community analysis shows that community measures (abundance, species richness, Shannon Wiener Diversity,

Table 8. Summary of Sediment Toxicity Test Results for Six Samples (Mean ± SD) from each of Three Study Areas in the Strait of Canso, NS, Canada.

Sample ID	Polydora % survival	Poydora weight (mg)	*E. estuarius* survival %	Microtox EC50	Sea urchin fertil-ization in 100% porewater	Total ammonia ug/g N	Sulfide ug/g	Redox mv
Nearfield	86.7 ± 21	1.17 ± 0.3	88.3 ± 6	1900 ± 840	52.0 ± 15	14.6 ± 4	28.8 ± 8	17.2 ± 7
Dumpsite	60 ± 28	0.70 ± 0.7	78.3 ± 8	1490 ± 300	47.0 ± 6	19.7 ± 12	61.9 ± 18	7.33 ± 18
Reference site	80 ± 13	0.61 ± 0.5	88.3 ± 8	1460 ± 560	37.4 ± 7	24.7 ± 11	54.2 ± 22	30.2 ± 20

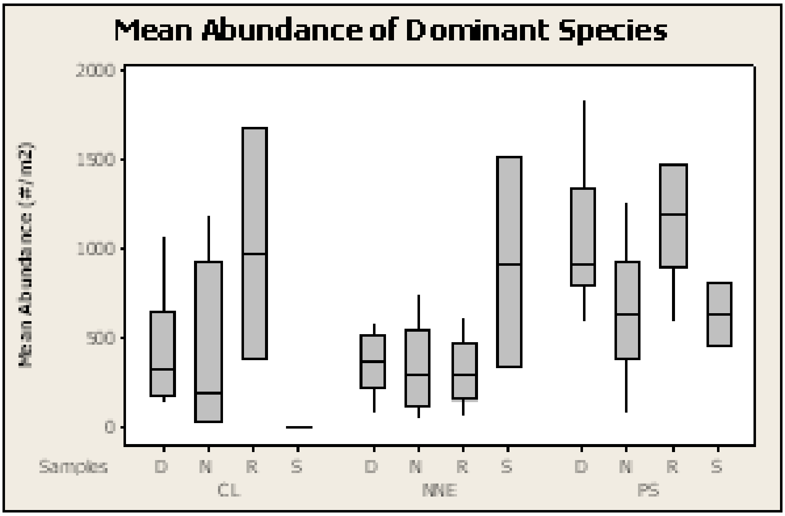


Figure 24. Mean Abundance of Dominant Species (CL — *Cossura Longocirrata*; NNE — *Ninoe Nigripes*; and PS — *Prionospio Steenstrupi*) at the Four Studied Areas (D — Disposal Site; N — Nearfield Site; R — Reference Site; and S — Outside Area).

Pielou's Evenness, and biomass) are similar between disposal and reference sites (Figs. 24 and 25) (Table 9) indicating that the communities at the disposal site are not under stress. No difference of the community measures including total organisms abundance and biomass was observed between the 1987 and 2004 benthic data, although the number of species is lower in 2004. Two species that were dominant in 1987, the polychaetes *Nereimyra punctata* and *Spiophanes kroyeri*, were not found in the 2004 samples. The smaller number of species and the absence of the two polychaete species in the 2004 samples may be an indication of the physical impact caused by the recent disposal activities occurred in 2000.

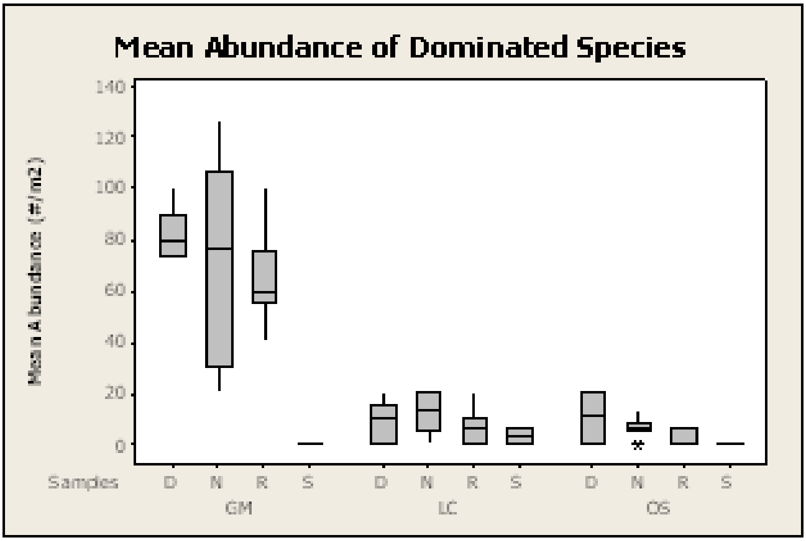


Figure 25. Mean Abundance of Dominant Species (GM — *Goniada Maculate*; LC — *Laonice Cirrata*; and OS — *Ophiura Sarsi*) at the Four Studied Areas (D — Disposal Site; N — Nearfield Site; R — Reference Site; and S — Outside Area).

Overall, in spite of a slightly elevated PAH concentration in one of the disposal site samples and the marginal (not statistically significant) toxic effect on the sea urchin and polychaete tests, the *in-situ* benthic data have confirmed that past disposal activities did not cause a significant impact at the disposal site. Dredged material disposal at the Canso common disposal site is not an annual event and materials disposed at the site are not high in contaminants. The monitoring program conducted at this site has accepted the hypotheses that disposal at the site has not caused significant adverse impacts.

5.3. *Black Point Dredged Material Disposal Site, Saint John, New Brunswick, Canada*

The Black Point disposal site, located off Saint John Harbour, New Brunswick, is the largest dredged material disposal site in Atlantic Canada. The site has been used annually since the early 1960s.[77] Annual volumes of dredged material placed at the site have ranged from approximately 100,000 cubic meters up to one million cubic meters. It is located in the harbor approaches and is affected by strong currents, eight-meter-high tides, and waves from winter storms. Based on the transport pattern in this area, it was initially predicted that the dredged material disposed of at the site would be quickly dispersed by the strong outwardly directed current.[78] However,

Table 9. Community Measures at all Studied Areas of the Canso Disposal Site. Based on Three Replicates (0.05 m^2) Per Station.

	Abundance (number/m^2)		Species per 0.05 m^2 Sample		Shannon-Wiener Diversity (Log10)		Pielou's Evenness		Simpson's Index		Biomass (grams/m2 wet Weight)	
Station	0	S.D.	0	S.D.	0	S.D.	0	S.D.	0	S.D.	0	S.D.
Nearshore												
A3	3647	1544	13	2.9	0.69	0.033	0.621	0.071	0.269	0.039	51.7	38.1
C3	1053	891	6	1.5	0.588	0.096	0.804	0.156	0.318	0.056	8.3	6.7
Disposal Site												
Dump 1	2080	1422.8	7	1	0.501	0.059	0.6	0.116	0.425	0.081	46.3	33.5
Dump 2	2720	1119.5	7.7	0.6	0.5	0.027	0.566	0.028	0.4	0.038	43.9	21.2
Dump 3	1520	904.2	6.7	2.1	0.5	0.031	0.627	0.105	0.405	0.056	5.8	5
Dump 4	3933.3	2435.3	6	1	0.482	0.021	0.627	0.072	0.389	0.042	21.3	15.8
Dump 5	1000	1128	9	1	0.749	0.143	0.793	0.186	0.234	0.1	92.3	119.4
Dump 6	1446.7	1270.6	8	1	0.651	0.146	0.726	0.181	0.305	0.123	31.1	4.7
Nearfield												
Near 1	1686.7	1436.4	8.3	2.5	0.587	0.02	0.658	0.113	0.334	0.03	46.4	39.4
Near 2	1400	87.2	8.7	0.6	0.523	0.037	0.558	0.024	0.398	0.025	21	11.2
Near 3	1353.3	674.5	7.7	2.1	0.527	0.073	0.61	0.128	0.382	0.047	13.3	10.5
Near 4	413.3	291.4	5	1.7	0.565	0.171	0.825	0.054	0.335	0.149	19.9	28.7
Near 5	886.7	421.6	7.3	0.6	0.635	0.09	0.737	0.129	0.318	0.113	21.8	10.2
Near 6	2780	1594	7	1	0.454	0.007	0.541	0.034	0.438	0.01	21.5	14.7
Reference												
Ref 1	1967	1052	7	1.5	0.566	0.039	0.662	0.054	0.346	0.022	46	35.7
Ref 2	3967	352	10	1.2	0.539	0.038	0.549	0.036	0.348	0.027	122.4	131.5
Ref 3	3040	2598	6	2.5	0.426	0.075	0.553	0.035	0.452	0.064	33.1	23.8
Ref 4	1887	1398.5	7	2	0.503	0.103	0.628	0.225	0.407	0.12	87	91.5
Ref 5	2093	925.7	7	1	0.465	0.033	0.552	0.013	0.474	0.023	56.6	33.9
Ref 6	2807	685.4	8	2	0.496	0.057	0.565	0.04	0.395	0.057	39.2	13.6

studies conducted at the disposal site indicated that past dumping has created an accumulation of dredged materials with a chemical imprint.[69,79]

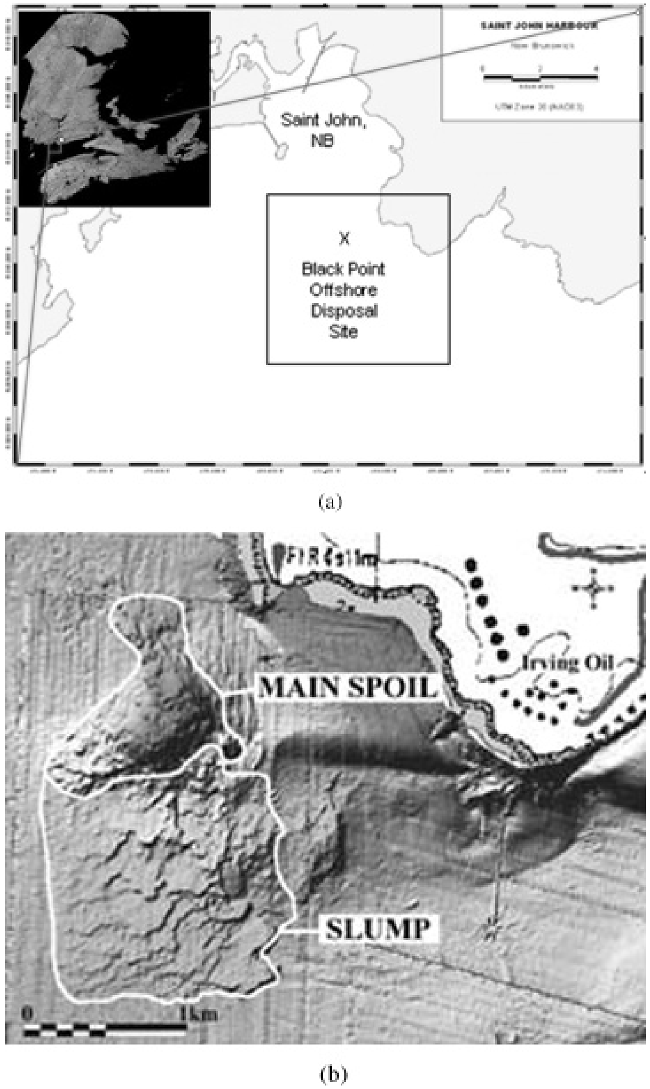
Between 1992 and 1994, Environment Canada conducted a three-year study at the site to assess the environmental alterations reported by the previous workers.[77] The study included the use of single-beam echo-sounders and side-scan sonar equipment, sediment chemistry, sediment toxicity studies, and benthic biota sampling. The results of the study indicated that, in spite of the high-energy environment of the disposal grounds, past disposal activities resulted in a significant buildup of dredged material and that the disposal activities had resulted in an appearance of a distinct benthic community at the disposal site.

In 1999, one year after the publication of the Environment Canada's monitoring guidelines, a more extensive physical monitoring survey was conducted at the site to reassess the accumulation of the dredged material.

From 2000 to 2005, geophysical survey including multibeam bathymetry, side-scan sonar, sub-bottom profiling, sediment physical and chemical analyses, and bottom pictures were collected from the site before and after disposal activities. The multibeam bathymetry data has shown that the disposal of dredge material has produced an irregularly shaped mound with a 1.5-km radius (Fig. 26). Historical bathymetry records had shown that the mound came to within 3 m of the water surface at low tide. A large slump, roughly 1.5 km by 1.5 km in size, had developed southward from the spoil mound due to the failure of the dumped material. Although material has accumulated at the site, a considerable portion of the material appears to have been transported away from the disposal site. Strong tidal currents of up to 60 cm/s were measured at the disposal site. Calculations show that fine sand sediment at the site was mobile more than 50% of the time in a tidal cycle.

Predictions from hydrodynamic and sediment transport models (Fig. 27) indicated that for uniform fine sand under tidally induced current conditions, the maximum total transport rate at the disposal site could reach 0.024 kg/m/s, with an eastward direction during flood and westward direction during ebb tides. Transport rates were generally less than 0.00027 kg/m/s in the inner harbor, but gradually increased to 0.0027 and 0.27 kg/m/s in the outer harbor and the interior Bay of Fundy, respectively. Outside the harbor, the direction of sediment transport was to the east and northeast during flood and to the west and southwest during ebb tides. The net transport, averaged over a tidal cycle suggested movement of sediment to the southwest in the inner harbor, and to the east and southeast in other areas of the modeled region. Superimposition of one-year storm waves could enhance the transport rates by one to two orders of magnitude.[19]

Figure 28 shows a series of images generated from multibeam bathymetry data from the disposal site. Figure 28(a) shows a color-coded multibeam bathymetry image of the seafloor at the Black Point disposal site in November 2004. The large



(a)

(b)

Figure 26. (a) Location of the Black Point Ocean Disposal Site in the Approaches to Saint John, NB. The Survey Area is shown on the Inset Map of Eastern Canada. (b) Shaded-Relief Image Created from Multibeam Bathymetry Data Collected Over the Disposal Site Showing the Main Disposal Pile and Material that has Slumped into Deeper Water. The Location of the Designated Offshore Disposal Site is Shown by the Black Rectangle.

spoils pile developed prior to 2001 is evident in the central portion of the image. In 2001, the disposal activity was relocated about 500 m west of the original site, resulting in the smaller accumulations of material visible west of the main pile. The acoustic backscatter intensity image of the Black Point offshore disposal site in November 2004 (Fig. 28(b)) shows the accumulation of fine sediments as low

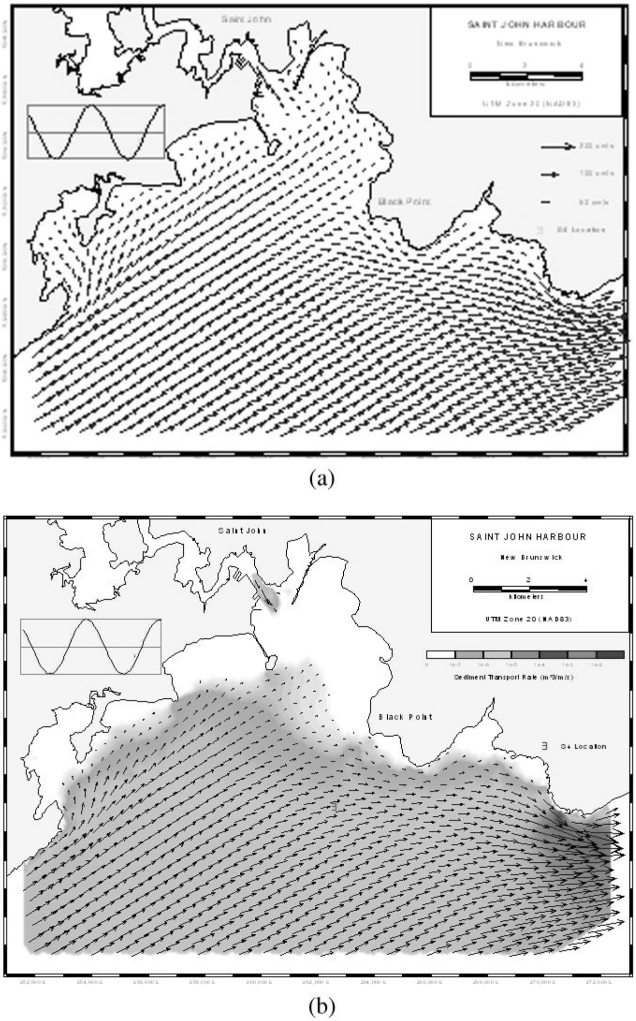
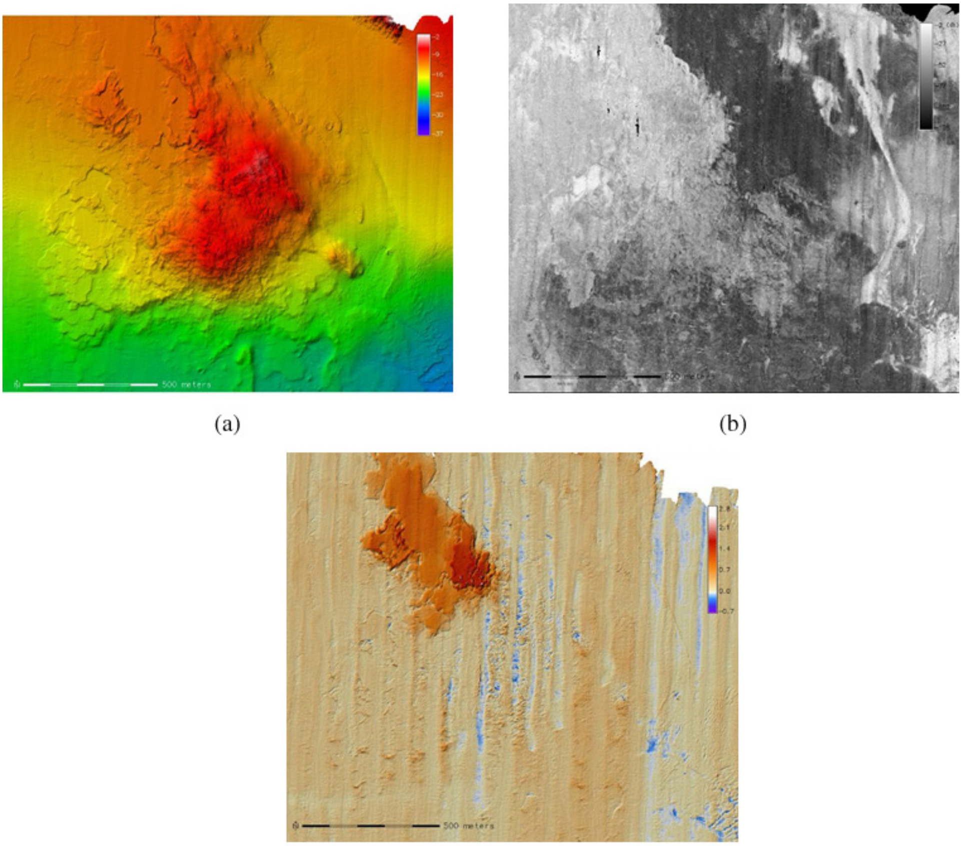


Figure 27. (a) Results of the Tidal Model for Currents at Peak Flooding Tide. (b) Results of Sediment Transport Model for Fine Grained Sand at Peak Flooding Tide Near the Black Point Disposal Site.

acoustic backscatter shown by the lighter colors. Areas of darker color show an increase in backscatter indicating the presence of coarse material such as gravel, boulders, and bedrock. The resolution of the multibeam bathymetry systems enables the calculation of difference surfaces from repetitive surveys. Figure 28(c) shows the accumulation of dredge spoils at the Black Point disposal site from July 2004 to November 2004 when 139,515 m^3 (scow measure) of materials were disposed at the site. A layer of the disposed materials, ranging from 1.0 to 2.5 m, have accumulated over the disposal site. An estimated 128,000 m^3 of materials were deposited, based on the calculated difference between the two multibeam bathymetry



(a)

(b)

(c)

Figure 28. (a) Multibeam Bathymetry Results from a November 2004 Survey of the Black Point Disposal Site. (b) Backscatter Intensity Derived from the Multibeam Bathymetry. (c) Difference in Elevation Between Multibeam Bathymetry Surveys Conducted in July 2004 and November 2004, Showing the Accumulation of Sediments During the Disposal Season. Color Bars are Shown for Each Plot.

surveys. A repeat survey in July 2005 showed a difference of 49,000 m^3 between the July 2004 survey, indicating that about 40% of the dredge materials remained after the winter.

The evidence of active bedforms in the area of the disposal site and of sediment transport from the site provide additional justification for addressing questions of off-site physical, chemical, and biological effects resulting from use of the site. To understand the significance of disposal related effects, the regional sediment fluxes must also be better understood. Specific issues that need to be addressed are:

- the physical and chemical characteristics and the volume of recently deposited sediments undergoing active transport to areas outside the disposal site boundaries,

- the directions and distances likely traveled of sediments moving from the disposal site,
- the effects on habitat outside the disposal site resulting from ongoing disposal activities, and
- the significance of the effects in relation to the overall movement of sediments in the study area.

Based on the physical monitoring results, which indicate a potential westward movement of dredged materials from the disposal site, a suite of benthic biota sampling stations was selected using a control-impact component involving stations west and east of the disposal site at different distances to determine if the disposal site was affecting adjacent communities. Community measures including abundance, biomass, number of species, and diversity did not detect major differences in the community at different distances west from the disposal site. Although elevated abundance of certain species, particularly the small polychaetes *Cossura longocirrata* and *Paraonis gracilis*, just beyond the edge of the disposal site boundary do suggest that there is enhancement activity because of the disposals.

Benthic biota collected at the disposal site are typically present in reduced abundance and number of species compared to adjacent areas. It is likely that the community at the disposal site has been continually affected by the physical impacts of the annual disposal activities at the site. The changes of physical characteristic of the dredged materials and the continued erosion of the materials at the site are also factors that influencing the establishment of a stable benthic community at the site.

Extensive sediment chemistry has been conducted on samples collected from the disposal site since 1992. The results indicate that, in general, heavy metals are not a concern in the sediments while PAHs and PCBs can be detected at some individual stations located at the disposal site. These sediments show toxicity as interpreted from the suite of sediment bioassays including the amphipod test (*Amphiporeia virginiania* and *Rhepoxynius abronius*), Microtox® Solid-phase, and sea urchin (*Lytechinus pictus*) tests. However, since sediment Eh, sulfide, and oxygen uptake data were significantly correlated with the end points of some of these tests, it was concluded that the anoxic nature of the sediments was probably responsible for the toxicity, not the contaminants. As a precaution, lobsters (*Homarus americanus*) were collected from the disposal site to asses the levels of PAHs and PCB in the tissues. The PAH and PCB concentrations in the cooked lobster meat and digestive glands were well below those reported for other major harbors in Atlantic Canada. These biological uptake results are consistent with tissue uptake study using blue mussel, *Mytilus edulis*, collected from the disposal site and the ethoxyresorufin 0-de-ethylase (EROD) and cyanoethoxycoumarin 0-de-ethylase (CN-ECOD) enzyme activity study using winter flounder caught southeast of the disposal site.

The Black Point disposal site is located in a nearshore high-energy area. Monitoring at this site has encountered several problems: (1) the site is continually affected by discharges from the Saint John River in the harbor, the strong current in the Bay of Fundy, and the movement of sediments from the upper bay and (2) disposal at the site has created a shallow water shoal at the coastline resulting in the formation of a potential lobster habitat. Continued use of the site requires proper management planning. Data collected from the monitoring program will be used to design a disposal site management plan to strategically place the dredged materials at the disposal site to avoid direct disposal impact on the lobster.

6. Conclusion

Dredged material disposal site-monitoring programs are conducted to assess the environmental impacts of disposal activities and to verify the decision made by environmental scientists and managers in permitting the disposal of the materials, the use of the disposal site, and the prescribed disposal activities. The program usually involves physical, chemical, and biological monitoring which, sometimes, can be a very expensive operation. To ensure cost effectiveness and consistency, the guidance documents are developed by some LC 72 Contracting Parties to guide the design of monitoring programs. In general, a tiered monitoring approach is recommended, starting with physical monitoring to delineate disposal areas and following by chemical and biological monitorings to assess the disposal impacts. Sometimes, however, when cost is an issue, it may be more effective to conduct these programs concurrently, especially at a site located far offshore, with deep water depth or in an area with extreme currents and tides. An effective monitoring program should have a clear purpose, a testable hypothesis, and cost-conscious design. Not all monitoring tools described in this chapter need to be used at each dumpsite, only those that are most suited to the issues identified at each site. The data collected by the program should be useful for making effective management decisions concerning the use of the dredged material disposal site.

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Chapter 16

RADIOACTIVE POLLUTION AND CONTROL

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Abstract

In this chapter, a brief introduction is presented to radioactive pollution and its management. Both natural and artificial sources of radiation are discussed with special attention to the relative importance of each source. In addition, radioactive isotope applications in tracing, radiography, insect control, food preservation, medical diagnoses, therapy, sterilization, and power generation are briefly described. Sources of radioactive pollution are identified including nuclear weapon tests, nuclear accidents, routine effluent release into the environment, and radioactive waste. Radioactive pollution prevention measures are presented including: treaties, regulations and standards, and technical methods to control pollution. The risk-informed regulatory process is briefly described, along with the technical methods to control radioactive wastes via site selection criteria, design of radioactive waste disposal facilities, and performance assessment.

Keywords: Radioactive pollution, radioactive waste, control measures, sources, regulations, radioactive applications, standards and disposal facilities.

1. Introduction

Radiation is omnipresent on Earth in the form of natural radiation. All living organisms are continuously exposed to radiation from a variety of sources. Scientific understanding of radiation and its effects on humans and the environment dates back almost a century to the pioneering work of Roentgen and Becquerel. Less than 40 years later, the first purified radioactive materials were produced by the Curies and within a decade of this discovery, scientists had split the atom. Since then, radiation has been used for diagnosing and treating medical problems, generating electricity, and in many other industrial, agriculture, and research applications.

Pollution is one of the major problems that face humanity; it occurs when a substance is released into the environment in a manner or in quantities that prevent the environment from effectively handling it, leading to detrimental effects on the ecosystem.[1] Protecting the environment from the effect of radioactive pollution has received a great deal of attention from both governments and individuals. Figure 1 represents the public perception of the importance of different pollution sources.[2]

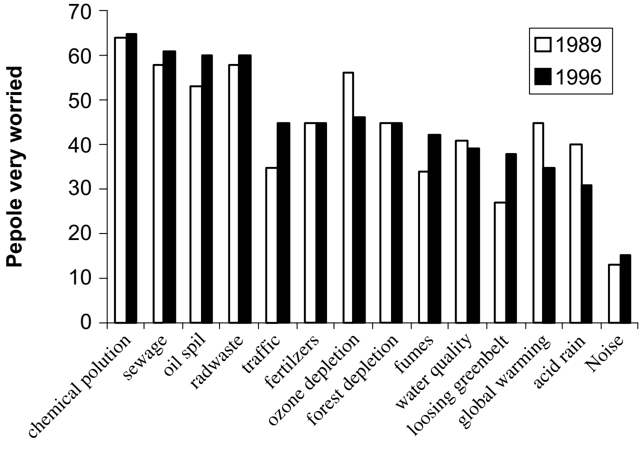


Figure 1. Public Awareness of the Importance of Different Pollution Sources.[2]

This chapter describes issues associated with radioactive pollution and control with special reference to the measures to control pollution from radioactive waste. Sources of radioactive pollution are identified and approaches to authorization and regulation to control radioactive pollution are presented.

1.1. *Radioactivity and Radiation*

The atom is the building block of matter; it consists of a small massive nucleus surrounded by a cloud of electrons. The nucleus is composed of positively charged protons and neutrally charged neutrons.[3] If the nucleus contains either excess neutrons or protons, the forces between these constituents will be unbalanced, leading to an unstable nucleus. An unstable nucleus will continually vibrate and will attempt to reach stability by undergoing radioactive decay. The major types of radiation emitted during radioactive decay include alpha particles, beta particles, gamma rays, and neutron radiation.[4]

1.1.1. *Alpha Particles* (α)

Alpha particles are energetic and positively charged helium nuclei consisting of two protons and two neutrons. The emission of these particles is comparatively rare in nuclides lighter than lead, but it is common for the heavier nuclides such as uranium-238, radium-226, and polonium-210. Even though these particles are highly energetic, their high mass leads to slow propagation through air, and they can be absorbed completely by paper or skin. Figure 2 represents the penetrating distance of different types of radiation.

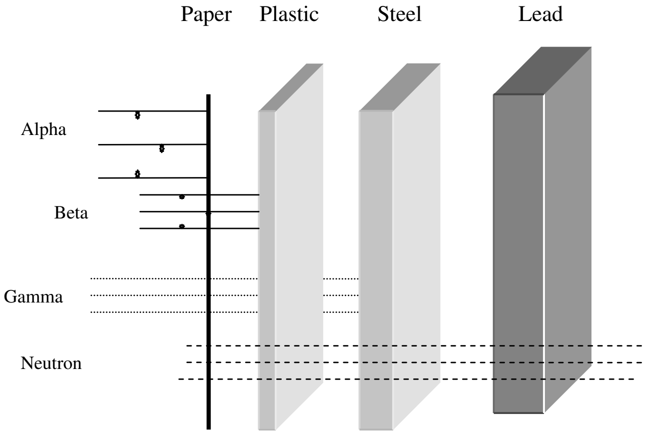


Figure 2. Penetration of Different Radiation Types.

1.1.2. *Beta Particles* (*β*)

Beta particles are fast-moving electrons emitted from the nucleus during radioactive decay. Two forms of beta decays exist. In the first, $^{+}\beta$ decay, one proton is transformed into a neutron and a positron and neutrino are emitted. In the second, $^{-}\beta$ decay, a neutron change into a proton and antineutrino are emitted. Human beings are exposed to beta particles from both artificial and natural radiation sources. Common radionuclides that produce significant beta radiation include tritium, carbon-14, and strontium-90. Beta particles are more penetrating than alpha particles: they can be completely absorbed by sheets of plastic, glass, or metal, as shown in Fig. 2. However, beta radiation is less damaging over equally traveled distances (Fig. 2).

1.1.3. *Gamma Rays* (*γ*)

Gamma rays are weightless packets of energy called photons, a form of high-frequency electromagnetic radiation. One source of gamma rays in the environment is naturally occurring potassium-40, which is present in the human body. Artificial sources include cobalt-60 and cesium-137. Gamma rays are very penetrating and only dense materials such as lead can provide good shielding from them (Fig. 2). Gamma rays can easily pass completely through the human body, with only a fraction absorbed by tissue.

1.1.4. *Neutron Radiation (n)*

Neutron radiation is a neutron emitted by an unstable nucleus, often fission and fusion. Neutrons are highly penetrating; when they interact with matter or tissue, they cause the emission of β and γ radiations (Fig. 2).

1.2. ***Radiation Units***

Absorbed dose (D) is defined as the amount of energy that ionizing radiation (ΔE) deposits in a unit mass of matter (Δm). It is expressed in gray (Gy), which is equal to 1 joule per kilogram.

Equivalent dose (H) is used to evaluate the relative biological effectiveness (RBE) of radiation to cause biological hazard. Biological hazard, for low doses of radiation, is defined as the risk of cancer mortality plus the risk of genetic effects. The International Commission on Radiation Protection (ICRP) has developed standard approaches for establishing the link between absorbed dose and effective dose. These standard approaches are periodically updated to reflect the best current understanding of the risk associated with radiation exposure. This means that the equivalent dose provides an index of the probability of hazard from exposure to different types of radiation. The SI unit of equivalent dose is Siveret (Sv), which is defined as the dose equivalent arising from an absorbed dose of 1 Gy. An older unit for the equivalent dose is rem (1 Sv = 100 rem), which is still widely used in several prominent nuclear countries, notably the United States and Russia.

The biological hazard of radiation to different parts of the human body varies from organ to another in both kind and magnitude. The effective dose is used to represent the overall biological hazard of radiation to a person as a single number; it is obtained by multiplying the equivalent dose by the organ weighting factor. The approach for determining the biological hazard of radiation is periodically updated.[5–7]

1.3. *Radiation Sources*

Radiation is a natural part of the environment. All living organisms, including human beings, are exposed to radiation daily. As a result, radiation exposures to radioactive pollution are incremental doses over the natural exposure. For the purposes of regulation of radioactive pollution, the incremental dose is often categorized as "artificial" radiation to distinguish it from "natural" background radiation. This distinction is adopted for the structure of this chapter, but it is noteworthy that there is no physical, chemical, radiological, or physiological differences between "artificial" and "natural" radioactivities. United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) estimates that the annual dose average over the population of the world at the surface (mean individual dose in a population) is about 2.8 mSv. Of this value, 85% is from natural sources. The exposure to radon decay products at home represents about half of this value and the exposure of patients to radiation from medical uses accounts for another 14%. The remaining exposures are from a variety of different sources.[8] There is a degree of variation in this estimated mean value due to local conditions, including variation in radon decay products at home, altitude at which the person lives, and personal habits. These variations lead to annual doses of 10 mSv or more for local regions.[8] The percentage due to different radiation sources for mean individual dose is illustrated in Fig. 3.

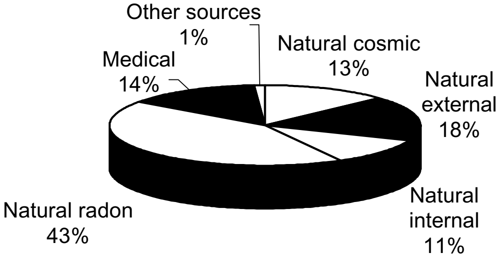


Figure 3. Relative Importance of Radiation Sources.[8]

1.3.1. *Natural Radiation*

Ionizing radiation and radioactive substances are natural and permanent features of the environment. Natural radiation comprises cosmic and terrestrial radiations.

1.3.1.1. Cosmic radiation

Cosmic radiation is produced when solar and galactic cosmic radiations collide with atoms of air (oxygen, nitrogen, and hydrogen) in the upper layers of the atmosphere to generate a complex set of secondary charged and uncharged particles, including protons, neutrons, pions, and lower-Z nuclei. These secondary particles generate more nucleons, producing a nucleonic cascade in the atmosphere. Figure 4 presents the contribution to dose equivalent rate from different cosmic ray components as a function of altitude.[9] From this figure, it is clear that at low altitude, muons represent the main source for exposure. As the altitude increases, the radiation exposure will be mainly due to neutrons, electrons/photons, and protons.

1.3.1.2. Terrestrial radiation

Terrestrial radiation is emitted by natural radioactive atoms present in natural materials such as soil, rocks, and clay. The predominant radioactive atoms in these materials include uranium, thorium, radium, and potassium. The decay of radium generates radon, a radioactive gas that is a high contributor to the overall terrestrial dose for many people.[10] When the building material or the surrounding soils

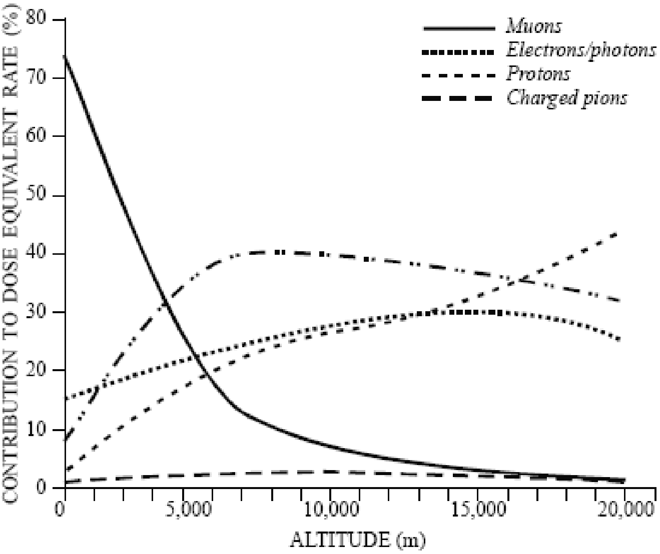


Figure 4. Components of the Dose Equivalent Rate Form Cosmic Rays in the Atmosphere.[9]

contain radium, radon gas produced from the radium can accumulate in the home; for homes with low air ventilation radon concentrations can be high and lead to significant radiation exposures.[11]

There are regions around the world where the concentrations of thorium, uranium, and other natural radionuclides in soil and beach sand exceed the average background levels by large margins. As examples, in the Brazilian coastal town of Guarapari and along the Kerala coast in India, the personal annual doses of members of the public may exceed the occupational annual dose limit of 20 mSv because of high levels of uranium and thorium in the local geology.[12–15]

1.3.2. *Artificial Radiation Source*

Artificial radiation sources are produced in a nuclear reactor or accelerator. These radiation sources are used in both medicine and industry. The main users of artificial radionuclides include medical facilities, such as hospitals and pharmaceutical facilities; research and teaching institutions; and nuclear fuel cycle facilities, such as power reactors, uranium mills, and fuel fabrication plants. Figure 5 illustrates the mean individual dose from different artificial and natural radiation sources.[3]

1.4. *Hazard of Radioactive Materials*

Radiation has sufficient energy to strip away electrons from atoms (creating two charged ions) or to break some chemical bonds. Living tissue can be damaged by ionizing radiation when it absorbs energy. The body attempts to repair the damage; however, sometimes the damage is of a nature that cannot be repaired or it is too severe or widespread to be repaired. Furthermore, mistakes made in the natural repair process can lead to cancerous cells. The biological effect of radiation can be classified according to the type of exposure.[16]

1.4.1. *Health Effects of Chronic Exposure*

Chronic exposure refers to long-term radiation exposure at a low to moderate level. This type of exposure leads to a probability of a health effect rather than to the certainty of a health effect; such conditions are known as *stochastic health effects*. The increased risk of developing cancer, cataracts, and genetic effects is considered among the possible stochastic effects. Genetic effects refer to damage to genetic material in a cell chromosome. Genetic effects can be ***somatic*** at which individual has experienced damage to some genetic material in the cell that could eventually cause the cell to become cancerous or ***hereditary*** in which the genetic effect is inherited or passed onto an offspring. These health effects have only been observed to occur at relatively high dose rates.

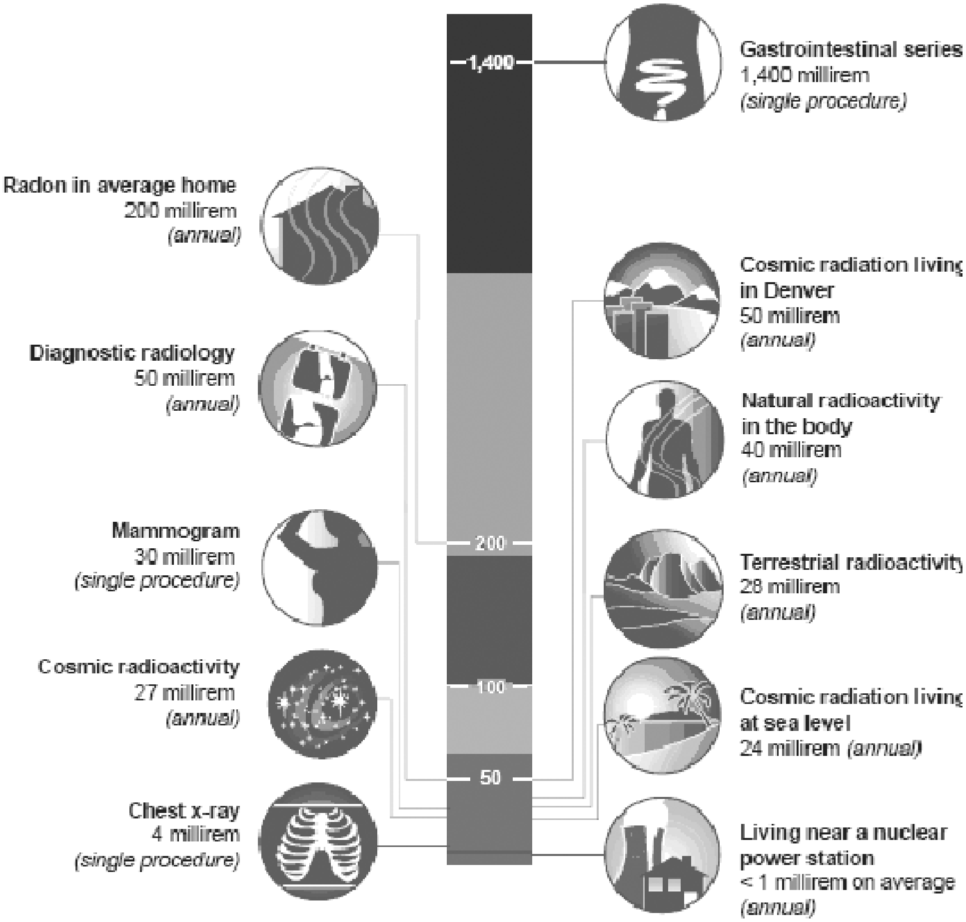


Figure 5. Relative Doses From Radiation Sources.[3]

The probability associated with stochastic health effects is too low to be observed at dose rates normally experienced by people, even for those who live in the regions of high background radiation. As a result, there is an assumption that the probability of health effects is linearly proportional to the absorbed dose, with no lower threshold below which no health effects occur. This assumption is commonly called the “linear, no threshold hypothesis” for risks associated with chronic exposure to radiation at very low doses. Increasing the exposure levels makes these health effects more likely to occur, but do not influence the type or severity of the effect. The “linear, no threshold hypothesis” has been the subject of some controversy in recent years, with some scientists suggesting the existence of a threshold below which radiation does not cause health effects, or is even beneficial (i.e. a hormetic effect at low absorbed dose).[17,18]

Table 1. Health Effect of Radiation Exposure.[17]

Exposure(rem)	Health Effect	Time to Onset
5–10	Changes in blood chemistry	
50	Nausea	Hours
55	Fatigue	
70	Vomiting	
75	Hair loss	2–3 Weeks
90	Diarrhea	
100	Hemorrhage	
400	Possible death	Within two months
1,000	Destruction of intestinal lining, internal bleeding, and death	1–2 Weeks
2,000	Damage to central nervous system, loss of consciousness, and death	Minutes, hours to days

1.4.2. *Health Effects of Acute Exposure*

Short-term and high-level exposure is referred to as acute exposure. Nonstochastic effects appear in acute exposure cases and become more severe, as the exposure increases. Acute health effects include burns and radiation sickness. Radiation sickness includes nausea, weakness, hair loss, skin burns, or diminished organ function. If the dose is fatal, death usually occurs within two months. Table 1 summarizes the health effects of radiation exposure.[19] For acute exposure at moderate dose rates the recovery is probable, includes the following possible effects: lowering of the white blood cell count, nausea, bacterial infections, vomiting, loss of appetite, reddening of the skin, diarrhea, fatigue, hair loss, and possible sterility. In a more severe exposure, the victim may suffer fever, abdominal pains, explosive diarrhea, internal bleeding, infection, shock, convulsions, coma, and ultimately death. Acute exposure is considered to be 250 mSv in 24 h.[20]

1.5. ***Technical Applications of Radioactive Materials***

1.5.1. *Radioisotope Tracing*

The concept of this technique is to replace one of the atoms in a molecule by a radioactive atom of the same element. Later, it is possible to trace the atom, as it undergoes physical or chemical transformations by observing the radiation emanating from it. To achieve the objectives of tracing, the physical form of the radiotracer is selected or manufactured so as to be consistent with the materials to be studied and its decay characteristics need to be appropriate.[21] Radioisotope tracing has proved to be effective in a number of different technical fields.

1.5.1.1. Environmental tracers

Radioisotope tracing plays an important role in detecting and analyzing pollutants, since even very small amounts of a radioisotope can easily be detected. Furthermore, through the use of rapidly decaying isotopes, experiments can be run leaving no residues in the environment. Nuclear techniques have been applied to a range of pollution problems including smog formation, sulfur dioxide contamination of the atmosphere, sewage dispersal from ocean outfalls, and oil spillage.[21]

1.5.1.2. Industrial tracers

By adding radioisotope tracers to materials used in various processes, it is possible to study the mixing and flow rates of a wide range of materials, including liquids, powders, and gases, and to locate leaks.[22] Tracers added to lubricating oils can help in measuring the rate of wear of engines, plant, and equipment. In addition, tracers have been used in plant operations to check the performance of equipment and improve its efficiency, resulting in energy savings and the better use of raw materials. In the oil and gas industries, unsealed radioactive solids (powder and granular forms), liquids, and gases are used to investigate or trace the movement of other materials, even within closed and otherwise inaccessible pipework and vessels.[23–26]

1.5.1.3. Fertilizers

Fertilizers labeled with radioactive isotopes, such as nitrogen-15 and phosphorus-32, provide a means of finding out how much fertilizer has been taken up by the plant and how much is lost.[27]

1.5.1.4. Application of radioisotopes tracing in water resources

Radioisotopes are used to trace and measure the extent of underground water resources. Such techniques provide important analytical tools in the management of existing supplies of water and in the identification of new and renewable sources of water. They provide answers to questions about origin, age, and distribution, the interconnections between ground and surface water, and renewal systems. The results permit informed recommendations for the planning and management of the sustainable use of these water resources. For surface waters, they can give information about leakage through dams, the dynamics of lakes and reservoirs, flow rates, and river discharge measurements and sedimentation rates.[28–30]

1.5.2. *Radiography*

Nondestructive testing (NDT) is commonly performed to provide quality assurance during engineering projects.[31] Gamma-emitting radioisotopes can be used to check welds of new gas and oil pipeline systems, with the radioactive source being placed inside the pipe and the film outside the welds. This is more convenient than employing X-ray equipment. Other forms of radiography (neutron radiography/autoradiography) can be used to gauge the thickness and density of materials, or to locate components that are not visible by other means (see Section 1.5.8).[32,33]

1.5.3. *Insect Control*

Crop losses caused by insects may amount to more than 10% of the total harvest worldwide; in some developing countries, the figure can be as high as 30%.[34] Chemical insecticides have for many years been humanity's main weapon in trying to reduce these losses, but they have not always been effective and can lead to toxic contamination of the foods they are intended to protect. Some insects have become resistant to the chemicals used and some insecticides leave poisonous residues on the crops. One solution has been the use of the sterile insects technique,[35] in which male insects are irradiated to sterilize them. Sterilized males are then released in large numbers in the infected areas. When they mate with females, no offspring are produced. With repeated releases of sterilized males, the population of the insect pest in a given area is drastically reduced.[36]

1.5.4. *Increasing Genetic Variability*

Ionizing radiation in plant breeding has been used for several decades to produce new genetic lines of sorghum, garlic, wheat, bananas, beans, avocado, and peppers; all of which are more resistant to pests and more adaptable to harsh climatic conditions.[37]

1.5.5. *Application of Radioisotopes in Food Preservation*

There is growing worldwide use of irradiation technology to preserve food. In almost 40 countries, health and safety authorities have approved irradiation of many kinds of food, ranging from spices, grains, and grain products to fruit, vegetables, and meat.[38] Following three decades of testing, a worldwide standard was adopted in 1983 by a joint committee of the World Health Organization (WHO), Food and Agriculture Organisation of the United Nations (FAO), and International Automic Energy Agency (IAEA).[39] In addition to reducing spoilage after harvesting, increased use of food irradiation is driven by concerns about food-borne diseases as well as growing international trade in foodstuffs, which must meet stringent standards of quality. Radiation is also used to sterilize food packaging. In the Netherlands, for example, milk cartons are freed from bacteria by irradiation.[40]

1.5.6. *Application of Radioisotope in Medicine*

Radiation and radioisotopes are used extensively in medicine, particularly diagnostically and therapeutically, for various medical conditions. Nuclear medicine mostly uses radioisotopes that emit gamma rays from within the body. It is estimated that about one out of every three hospital patients benefits in some way from the use of nuclear medicine.[41]

1.5.6.1. Medical diagnosis

Radioisotopes are an essential part of diagnostic treatment. In combination with imaging devices and computers, they are also used to study the dynamic processes taking place in the various organs. The procedure of medical diagnoses is based on giving a radioactive dose to the patient, then monitoring the activity in the studied organ. The organ can then be illustrated either as a two-dimensional picture or, with a special technique called tomography, as a three-dimensional picture. The most widely used diagnostic radioisotope is technetium-99 m, with a half-life of 6 h, and which gives the patient a very low radiation dose. Such isotopes are ideal for tracing many bodily processes with the minimum of discomfort and dose for the patient. They are widely used to indicate tumors and to study the heart, lungs, liver, kidneys, blood circulation and volume, and bone structure.[42,43]

A major use of radioisotopes for diagnosis is in radioimmunoassays for biochemical analysis. They can be used to measure very low concentrations of hormones, enzymes, hepatitis virus, some drugs, and a range of other substances in a sample of the patient's blood. The patient never comes in contact with the radioisotopes used in this diagnostic test.

1.5.6.2. Therapy

The uses of radioisotopes in therapy are comparatively few, but important. Iridium-192 implants in the form of a wire are used to give precise doses to limited areas. Iodine-131 is used to treat the thyroid for cancer and other conditions. Some cancers are treated using gamma rays from an external cobalt-60 source, others using internal beta radiation. A new treatment uses samarium-153 complexed with organic phosphate to relieve the pain of secondary cancers lodged in bone.[44,45]

1.5.6.3. Sterilization

Many medical products today are sterilized by gamma rays from a cobalt-60 source.[46] Since this technique does not require the application of heat, it is widely used in sterilizing a range of heat-sensitive items such as powders, ointments, and solutions and biological preparations such as bone, nerve, and skin used in tissue grafts. Medical products sterilized by radiation also include disposable syringes, cotton wool, burn dressings, surgical gloves, heart valves, bandages, plastic and rubber sheets, and surgical instruments.[47]

1.5.7. *Application of Radioisotope in Smoke Detectors*

One of the most common uses of radioisotopes is in smoke detectors. These contain a small amount of americium-241, which is a decay product of plutonium-241 originating in nuclear reactors. The americium-241 emits alpha particles, which ionize the air and allow a current to flow between two electrodes. If smoke enters the detector, it absorbs the alpha particles and interrupts the current, setting off the alarm.

1.5.8. *Application of Radioisotope in Instruments*

Gauges containing radioactive sources are in wide use in all industries where levels of gases, liquids, and solids must be checked.[48] These gauges are most useful where heat, pressure or corrosive substances, such as molten glass or molten metal, make it impossible or difficult to use direct contact gauges. Radioisotope thickness gauges are used in the making of continuous sheets of material including paper, plastic film, metal, etc., when it is desirable to avoid contact between the gauge and the material.

Density gauges are used where automatic control of a liquid, powder, or solid is important, for example, in detergent manufacture. Radioisotope instruments have three great advantages[48]:

1. Measurements can be made without physical contact with the material or product being measured.
2. Very little maintenance of the isotope source is necessary.
3. The cost/benefit ratio is excellent — many instruments pay for themselves within a few months through the savings they allow.

1.5.9. *Power Sources*

When radiation is absorbed, the radiated energy appears in the form of heat. A portion of this heat can be converted into electrical energy. Devices based on this principle are used for heart pacers and to power navigation beacons and satellites. The decay heat of plutonium-238 has powered many space vehicles and enabled voyager to send back pictures of distant planets. Plutonium-238 powered the Cassini space probe on its way to Saturn.[49]

2. Radioactive Pollution

The many applications of radioisotopes for industrial, medical, and power production inevitably lead to the releases of some of those radioisotopes into the surrounding environment. These releases may be categorized as either (1) planned releases that are a normal part of the application of the technology or (2) unplanned or uncontrolled releases associated with accidents. In the case of routine releases to the environment, each practice is evaluated to assure that the benefits outweigh the detriment,[50] and that the health risks from the routine release are negligible. For accidental releases, governmental authorities use action levels to determine when cleanup activities are required to protect the public health.[50]

Radioactive pollution arises from the discharge of radionuclides to the environment by nuclear power facilities, military establishments, research organizations, hospitals, and general industry. In addition, historical tests of nuclear weapons in the atmosphere and underground, nuclear and radioactive accidents, and the deliberate discharge of radioactive wastes from nuclear and other installations represent sources for radioactive pollution. Such radionuclides have the potential to find their way from air and water onto the ground and into the food chain.

2.1. *Sources of Radioactive Pollution*

2.1.1. *Military Activities and the Production, Testing, and Use of Nuclear Weapons*

The manufacture of nuclear weapons involves handling, transport, and storage of large quantities of radioactive materials. Weapons testing may involve the release of fission and activation products into the environment, and has in the past involved the deliberate dispersal of radioactive materials in the environment through atmospheric weapons testing.[51]

Atmospheric testing of nuclear weapons has been the most significant worldwide radioactive pollution source. This practice continued from 1945 to 1990 in various countries. The annual deposition of strontium-90 in different countries from the atmospheric tests is illustrated in Fig. 6. From this figure, it is obvious that the contribution of the test program of the United States dominated before 1958; then from 1959 to 1967, the dominant deposition of ^{90}Sr was contributed from the Soviet Union program. From 1968 to 1988, the deposition was primarily from the Chinese tests.

Nuclear weapons tested above the ground propelled a variety of radionuclides from tritium to plutonium into the upper atmosphere. From there, the radionuclides transferred slowly to the lower atmosphere and then to the earth's surface. Globally, the most important radionuclides from this source in terms of human exposure are now carbon-14, strontium-90, and cesium-137. Minute quantities of these are ingested with food and drink. Residual activity from radionuclides in the ground that emit gamma rays also causes a slight increase in human exposure. Internal and external irradiations contribute about equally to global average effective dose of 0.005 mSv in a year. This compares with a peak of more than 0.1 mSv in 1963. The global collective dose from weapon tests fallout is now about 30,000 man Sv annually, assuming a world population of 6,000 million.[51]

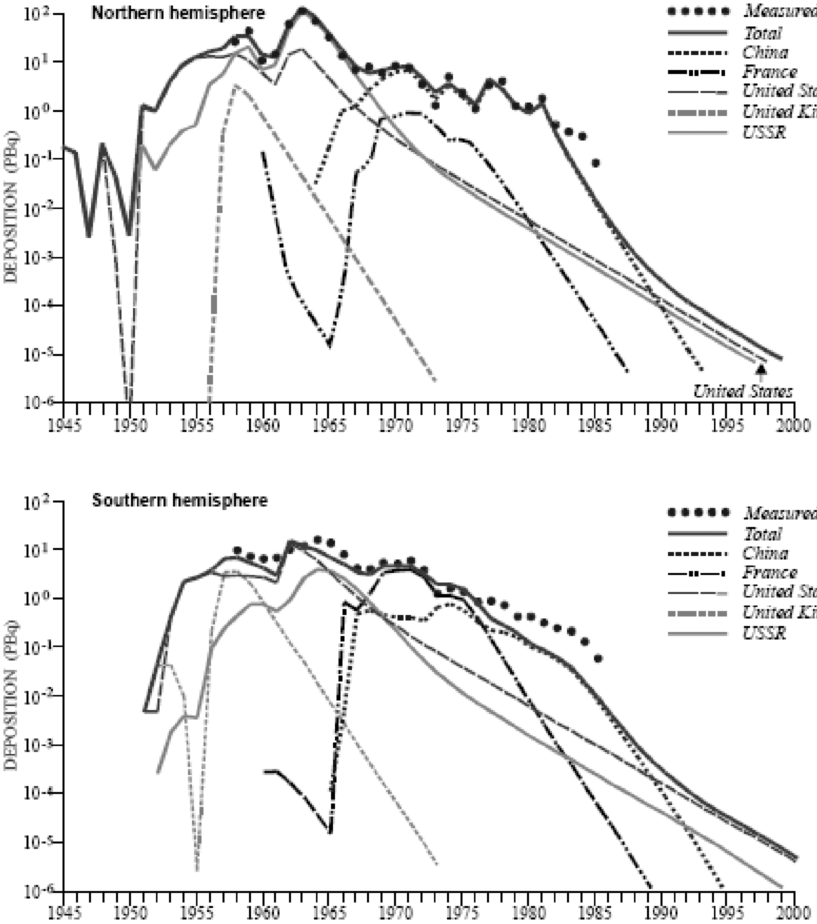


Figure 6. Components of Strontium-90 Deposition From Test Programs of Countries Calculated From Fission Yields of Tests With the Atmospheric Model.[50]

In addition to atmospheric testing, nuclear weapons were also tested underground in several countries, with the most recent of these tests conducted in 1998. Underground testing resulted in only localized releases of radionuclides to the environment, and significantly reduced the exposure of the population compared to atmospheric testing. Figure 7 represents the number and yield of underground and atmospheric nuclear weapon tests.[53]

Besides nuclear weapons tests, the military fuel cycle has also resulted in the releases of radioactive materials to the environment. These releases are localized at the manufacturing facilities in countries constructing nuclear weapons, but have led to significant local contamination in locations such as Hanford, the United States, and Chelyabinsk, Russia.[54,55]

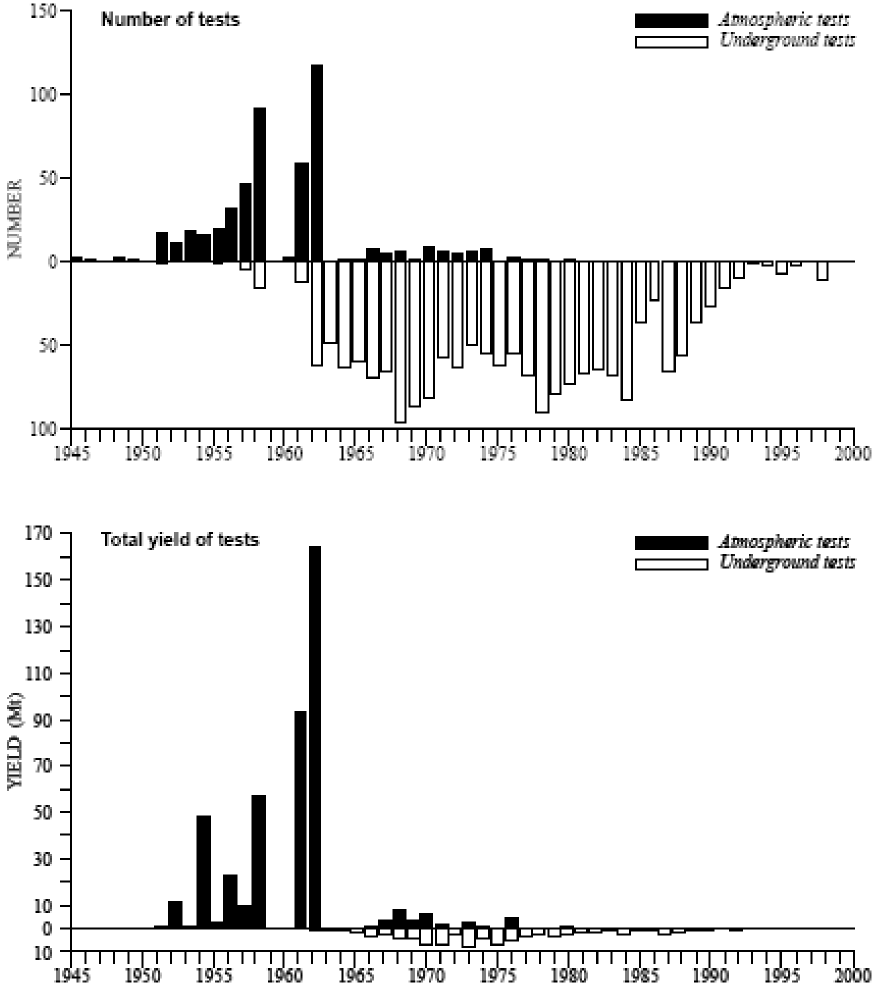


Figure 7. Test of Nuclear Weapons in the Atmosphere and Underground.[50]

Depleted uranium has been used for projectile weapons during recent conflicts.[56] This weapon is in a concentrated metallic from, and there have been concerns expressed by the public about elevated levels of radioactivity in the environment due to spent munitions. Depleted uranium can potentially have both chemical and radiological toxicities to the kidneys and the lungs. Health effects are determined by the physical and chemical natures of the depleted uranium to which an individual is exposed, and to the level and duration of exposure.[57,58] The WHO calculated the amount and fate of depleted uranium deposited at an "average" attack site in Kosovo.[59] The calculations showed that, if all the depleted uranium munitions expended during an attack remained within a kilometer, the increase of uranium in the soil would be 5%. The contribution of depleted uranium from military use to background radiation dose in Kosovo is within the natural variations found for background levels.[59]

2.1.2. *Nuclear and Radioactive Accidents*

Accidents involving the releases of radionuclides to the environment have occurred in power production, nuclear weapons production, and industrial applications. In the course of nuclear weapons production and transport, there have been several severe accidents resulting in considerable contamination.[60–63] These include Windscale Pile 1 (1957), Kyshtym (1957),[55] Palomares (1966), and Thule (1968).

The 1986 fire at the Chernobyl reactor led to widespread contamination. Airborne material was dispersed throughout Europe from the site in Ukraine. As the contaminated air spread throughout Europe and beyond, local weather conditions largely determined where the radionuclides were to fall. Rainfall caused more radionuclides to be deposited in some areas rather than others. The impact of the accident on the workers has involved an enormous number of people. The accident caused the deaths within a few days or weeks of 30 power plant employees and firemen (including 28 deaths that were due to radiation exposure), brought about the evacuation of about 116,000 people from the areas surrounding the reactor during 1986, and the relocation, after 1986, of about 220,000 people from Belorussia, the Russian Soviet Federated Socialist Republic (RSFSR) and the Ukraine.[64–67] However, health effects associated with the radioactive contamination itself have been small, and the primary health-related impacts of the accident were associated with the stress from the relocations.[68]

On 11 March 2011, a massive earthquake followed by a tsunami hit Fukushima Daiichi nuclear power station. The site includes 4 boiling water reactors (BWR) with Mark 1 containment. The site lost its external power due to the earthquake. It was planned to use emergency diesel generators to secure the power supply. However, these generators, the electric equipment rooms, and outdoor sea water pumps were submerged by the tsunami and lost their ability to function. As a result, the reactor experienced a loss of coolant, and it was necessary to inject seawater to maintain temperatures. Hydrogen accumulated in the containment building leading to explosions at the site. To achieve cold shutdown conditions, large amounts of water were injected into the reactors. The injected water leaked through the reactor pressure vessel and primary contaminant vessel into the basement of the building. Efforts were directed to minimize the accumulation of water, and to reuse the contaminated water for injection into the reactor. The radiation dose rate increased rapidly during the accident, and it has been in a stable declining trend since. It is now approximately at background levels. Three phases are planned for site remediation. The first is the removal of fuel from the spent fuel pools. The second is envisioned to take place over 10 years and aims to remove the fuel debris. The third stage comprises decommissioning, which is expected to end within 30 to 40 years.[69]

An accident involving loss of control of spent radiation sources occurred in Goiânia (1987). Four people died and six received doses of a few Gy from an abandoned and ruptured highly radioactive Cs-137 source.[70]

2.1.3. *Routine Release of Effluent to the Environment*

Nuclear electrical generation has grown steadily from the start of the industry in 1956. The relatively rapid rate of expansion that occurred from 1970 to 1985, an increase in energy generation of more than 20% per year, slowed to a pace averaging just over 2% per year from 1990 to 1996.[71] At the end of 1997, there were 437 nuclear reactors operating in 31 countries. The total installed capacity was 352 GW and the energy generated in 1997 was 254 GW. It is projected that nuclear energy will continue to supply about 17% of the total electrical energy generated in the world, as at present, or possibly a few percentage less. The nuclear fuel cycle includes mining and milling of uranium ore and its conversion to nuclear fuel material; the fabrication of fuel elements; the production of energy in the nuclear reactor; the storage of irradiated fuel or its reprocessing, with the recycling of the fissile and fertile materials recovered; and the storage and disposal of radioactive waste (Fig. 8).[72] At each stage of the nuclear fuel cycle, a variety of radionuclides are released in the form of liquid, gas, or solid particles. The nature of the effluent depends on the particular operation or process. Normalized releases and the resultant doses from each stage are presented in Fig. 9.[72]

2.1.3.1. Nuclear fuel cycle

Various stages of the nuclear fuel cycle and the operation and decommissioning of nuclear reactors all have the potential to create contaminated sites. The contamination may include mill tailings; spillage of ore end product at the mine and in transport; waste from enrichment and fuel fabrication operations; fission product and actinide waste streams from reprocessing of fuel elements; radioactive effluents from normal operations of nuclear power plants; wastes produced during decommissioning of reactors; and major releases under accident conditions.

2.1.3.2. Production and use of radioactive substances for medical, research, or industrial purposes

Radioactive materials have been used widely since their discovery for a variety of scientific, medical, and industrial uses. In some cases, either through ignorance, carelessness, or accident, sites have been left contaminated with residues of the

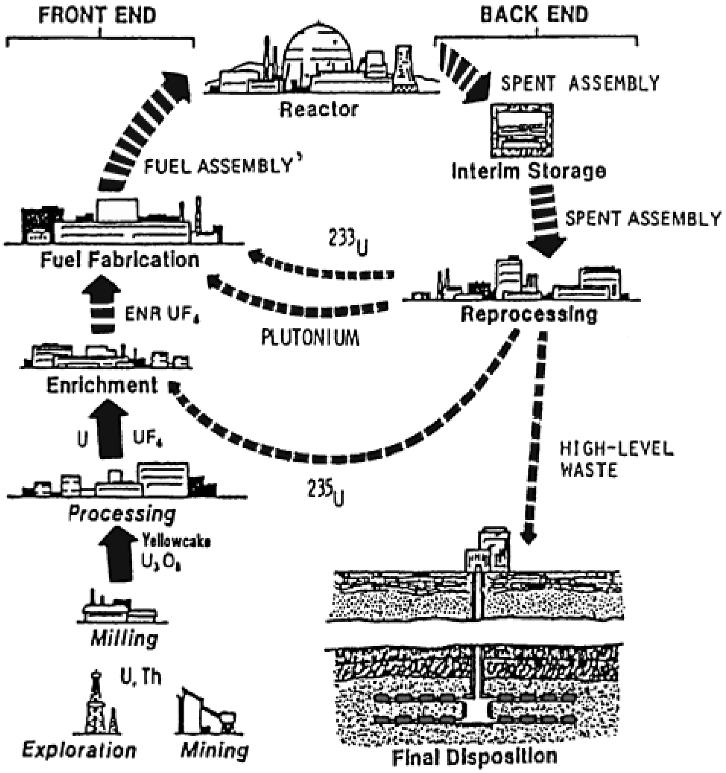


Figure 8. Nuclear Fuel Cycle.

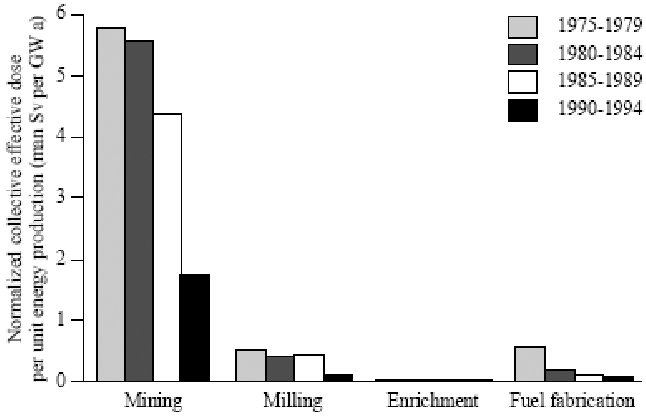


Figure 9. Normalized Collective Effective Dose Per Unit Energy Production for Mining, Milling, Enrichment, and Fuel Fabrication.[66]

Table 2. Activities That May Lead to NORM Contaminated Residues and Sites.[75]

Mineral Ores and Extracted Materials		Other Processing/Manufacturing
Copper	Titanium	Water treatment
Aluminum (bauxite)	Tungsten	Sewage treatment
Fluorspar	vanadium	Spas
Gypsum	Zircon	Paper and pulp
Iron	Coal (and coal ash)	Ceramics manufacture
Molybdenum	Oil and gas	Paint and pigment manufacture
Phosphate	Geothermal energy	Metal foundry
Phosphorous	Uranium and thorium	Optics
Potassium		Incandescent gas mantles
Precious		Refractory and abrasive sands
Rare earth		Electronics manufactures
Tin		Building materials

operations. Such sites include factories where radium was used in luminescent paint and thorium was used in thorium-coated gas mantles.[73]

2.1.4. *Radioactive Waste and Contaminated Areas With Naturally Occurring Radioactive Material (NORM) and Technically Enhanced NORM (TE-NORM)*

Because uranium and thorium are present in many ores containing other useful minerals, the mining of these ores and the processing to recover materials such as copper, gold, niobium, coal, and monazite will generally produce waste streams containing significant amounts of radioactivity. These have the potential to result in unacceptably contaminated sites.

Large areas have been contaminated in various parts in the world with radionuclides as a result of various human activities. Table 2 illustrates some activities that lead to NORM and TENORM contaminated residues and sites.[74,75] In cases where the level of contamination is high, measures might be needed to ensure that the area is safe to people to live or use for other purposes. For small areas, it might be possible to do this by removing contaminated soil and other materials, but for large areas, the amount of material would be too large. Other ways of protecting people include restriction on access to or use areas.

2.2. *Prevention of Radioactive Pollution*

Governments are responsible for protecting the public and environments; the manner in which this responsibility is implemented varies from country to country. In this section, the international agreements and declarations approved from various countries are presented, and the guidelines to authorize and regulate radioactive effluent release is summarized. The technical methods to control the pollution are also reviewed.

2.2.1. *International Agreements and Declarations*

Radioactive effluent releases are subject to several international agreements and declarations, which may impose obligations on national policies and procedures. The aim from these agreements and declarations is to prevent the occurrence of radioactive pollution from effluent release. The International Convention on Nuclear Safety[76] and the Joint Convention on the Safety of Spent Nuclear Fuel Management and on the Safety of Radioactive Waste Management,[77] which are signed and ratified by numerous united nation member states, are examples of such agreements and declarations. Member states of the European Community are legally bound by the provisions of the Euratom treaty[78] and the discharge in the north-east Atlantic is controlled by the OSPAR convention.[79]

2.2.1.1. International convention on nuclear safety[72]

The Convention on Nuclear Safety was adopted in 1994 and entered into force in 1996. The nuclear safety targets are contained in the articles of the convention and the mechanism for improving safety is through the peer pressure exerted upon each other by the contracting parties at the regular review meetings. With regard to the controlled release of effluents from nuclear power plants, Article 15 (Radiation Protection) of the convention requires that:

> Each contracting party shall take appropriate steps to ensure that in all operational states the radiation exposure to the workers and the public caused by a nuclear installation shall be kept as low as reasonably achievable and that no individual shall be exposed to radiation doses which exceed prescribed national dose limits.

2.2.1.2. Joint convention on the safety of spent nuclear fuel management and on the safety of radioactive waste management[77]

Several articles of the joint convention address issues related to discharges. The principal reference is in Article 24 (Operational Radiation Protection); Part 2 requires that:

> Each contracting party shall take appropriate steps to ensure that discharges shall be limited:
>
> (i) to keep exposure to radiation as low as reasonably achievable, economic and social factors being taken into account and
> (ii) so that no individual shall be exposed, in normal situations, to radiation doses which exceed national prescriptions for dose limitation which have due regard to internationally endorsed standards on radiation protection.

In this context, discharges are defined as planned and controlled releases into the environment, as a legitimate practice, within the limits authorized by the regulatory body, of liquid or gaseous radioactive materials, that originate from regulated nuclear facilities during normal operation. However, in addition, under Articles 6 and 13 on siting, each contracting party is required to

> "consult contracting parties in the vicinity of such a facility, insofar as they are likely to be affected by that facility, and provide them, upon their request, with general data relating to the facility to enable them to evaluate the likely safety impact of the facility upon their territory."

2.2.1.3. Euratom treaty[78]

At the time of signature of the Euratom Treaty in 1957, its main objective was to contribute to the raising of the standard of living in the Member States and to the development of relations with the other countries by creating the conditions necessary for the speedy establishment and growth of nuclear industries. This role is achieved by conferring to the Community far reaching competence to ensure the availability of nuclear materials for civil purposes (ownership of fissile material, safeguards), access to research and technical information, and investment funds. In addition, the development of nuclear industry should be conditioned by the establishment of uniform safety standards to protect the health of workers and the general public, the application of which shall be ensured by the community. Article 2 (b) of the EURATOM treaty provides for establishment within the community of

> "... uniform safety standards to protect the health of workers and of the general public"

and for the community to

> "... ensure that they [the standards] are applied."

Article 37 Euratom

> "... provide the commission with such general data relating to any plan for the disposal of radioactive waste in whatever form as will make it possible to determine whether the implementation of such plan is liable to result in the radioactive contamination of the water, soil or airspace of another Member State."

2.2.1.4. OSPAR convention for the protection of the marine environment of the North–East atlantic[79]

During the ministerial meeting of OSPAR in Sintra, Portugal, in 1998, the ministers and the member of the European Commission emphasized that:

> Our commitment to take all possible steps to achieve our overall objective for the protection of the marine environment of the North East Atlantic of preventing and eliminating pollution, protecting human health and ensuring sound and healthy marine ecosystems, and commit ourselves to pursuing this goal through the following actions to produce a sustainable approach

> to the marine environment of the OSPAR maritime area and thus protect this inheritance for the new millennium.

In addition, the ministers reemphasized, "the clear commitments to the application of the precautionary principle and the polluter-pays principle and to the identification of Best Available Techniques (BAT) and Best Environmental Practice (BEP), including, where appropriate, clean technology." This statement includes a part on radioactive substances, emphasizing the willingness of the ministers to ensure the implementation of the above-mentioned Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) strategy with regard to radioactive substances.

2.2.1.5. The Rio de Janeiro conference and the precautionary principle[80]

The Rio Declaration on Environment and Development states in Principle 15 — the Precautionary Approach:

> In order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation.

In the framework of Agenda 21, the precautionary principle is to be applied in cases of potential irreversible impacts on the environment with relative high consequences (implying that these consequences are unacceptable).

2.2.2. *Radioactive Effluent Discharge Standards*

2.2.2.1. IAEA safety standards

IAEA issues safety standards covering nuclear, radiation, transport, and waste safeties. Concerning the control of discharge, the principal requirements are contained in the International Basic Safety Standards for Protection (BSS) against Ionizing Radiation and for the Safety of Radiation Sources.[81] The BSS translates the basic radiation protection recommendations of the International Commission on Radiation Protection (ICRP) into regulatory form. The essential requirement is that:

> Registrants and licensees shall ensure that radioactive substances from authorized practices and sources not be discharged to the environment unless: discharge should be within the discharge limits authorized by the Regulatory Authority; discharges should be controlled; public exposures committed by the discharges should be limited; and control of the discharges should be optimized in accordance with the Principal Requirements of the Standards.

Detailed guidance on setting discharge authorizations is contained in a safety guide named "Regulatory Control of Radioactive Discharges to the Environment."[82]

This safety guide outlines the responsibilities of the regulatory body and of the organization intending to discharge radioactive material, sets out the steps to be followed in setting a discharge authorization for a new practice, and gives advice on actions to be taken in cases of noncompliance and on the procedures to be followed for existing discharge practices.

2.2.2.2. EU basic safety standards directive (96/29 Euratom)

The revised Radiation Protection Ordinance sets a dose limit of 1 mSv for members of the general public. Additional limits for doses resulting from radioactive discharges and emissions from nuclear installations are specified for aerial and liquid releases each: individual effective dose and partial body dose for gonads, uterus, and red bone marrow should not exceed 0.3 mSv/y and partial body dose of all organs and tissues unless under 2 and 4 should not exceed 0.9 mSv/y where the partial body of bone surface and skin should not exceed 1.8 mSv/y.

2.2.2.3. National regulations

Recently, several countries have evaluated their regulations regarding radioactive effluent releases from nuclear installations. This section will presents some national regulation in different countries.

2.2.2.3.1. *China*

The Law on the Prevention and Control of Radiation Pollution of the People's Republic of China has went into effect on October 1, 2003. The purpose of this law is to prevent and control radioactive pollution, protect the environment, ensure human health, and promote the development and peaceful use of nuclear energy and technology. The law establishes pollution and control measures for "radioactive pollution discharged in the course of site selection, construction, operation, and decommissioning of nuclear installations and in the cause of development and utilization of nuclear, technology, uranium (thorium) and accompanying radioactive mines in the territory of the People's Republic of China and in the territorial waters under its jurisdiction." Article 3 presents an important aspect of this law as it stipulates that, when concerning the prevention and control of radioactive pollution, the State should apply the principles of putting prevention first, combine prevention and control measures, exercise rigorous control, and give priority to safety.[83]

2.2.2.3.2. *Belgium*

The general regulation for protection of the population, workers, and the environment against the risk of ionizing radiation (Royal Decree of 20.07.2001) includes provisions limiting the exposure of the public to 1 mSv/y.

2.2.2.3.3. *France*

Decree 95-540 of May 4, 1995, and an Order dated November 26, 1999, set out technical directions concerning the discharge limits and method of sampling from discharges released by “Basic Nuclear Installations (BNI)”; the creation of these installations and their discharges both being subject to specific authorization.

2.2.2.3.4. *Japan*

The national standards of radiation protection for a nuclear installation are provided in the Law for the Regulation of Nuclear Source Material, Nuclear Fuel Material and Reactors, the Electricity Utilities Industry Law, etc. and related government ordinances, ministerial orders, and notifications based on these laws and guidelines. The recommendations of the ICRP are given due consideration and are incorporated into national legislation and regulations. The ICRP 1990 recommendation on radiation protection was incorporated into them in April 2001, after revision of related ministerial orders and notifications.

2.2.2.3.5. *United States*

Nuclear Regulatory Commission (NRC), under the Code of Federal Regulations (CFR), established revised requirements to its standards for protection again radiation.[84] NRC stated that:

> Due to the practice of maintaining radiation exposures ALARA (“as low as reasonably achievable”), the average radiation dose to occupationally exposed individuals [was] well below the limits in either the previous or amended 10 CFR Part 20 and also below the limits recommended by the ICRP.

In addition, NRC stated that:

> Until the final ICRP recommendations are published, and the need for further revisions in NRC regulations established, the Commission believes it would be advisable to proceed with the promulgation of the proposed dose limits [of 5 mSv per year], rather than deferring the dose reductions that are already associated with [its] amendments to Part 20.

As a result of the application of the ALARA philosophy to effluent release standards in Appendix I to 10 CFR Part 50 for nuclear power reactors and the U.S. Environmental Protection Agency’s (EPA’s) 40 CFR Part 190 for the uranium fuel cycle, dose from radioactive effluents from the fuel cycle were already much less than 1 mSv per year standard in the final rule. The 1 mSv per year remains as the level recommended by the ICRP in 1985 as the principle dose limit for members of the general public. More recently, in 1996, 10 CFR 20.1101 required an additional ALARA value for air emissions from licensed facilities that require that the individual member of the public likely to receive the highest dose will not receive in excess of 0.1 mSv per year from air emission. Failure to meet this requirement requires the licensee to submit a written report to the regulatory authority (NRC or the Agreement State). This change in regulation eliminated dual regulation of air emissions that had been previously regulated by both NRC and EPA.

2.3. *Risk Informed Regulatory Process and Importance Measure*

The fundamental objective of all nuclear safety regulatory bodies is to ensure that nuclear facilities are operated in an acceptably safe manner. Regulatory body should strive to ensure that its regulatory decisions are technically sound, consistent from case to case, and timely. The basic elements of regulatory decision-making process include clear definition of the issue, assessment of the safety significance, determination of laws, regulations or criteria to be applied, collection of relevant information and data, judgment the expertise and the resources needed, agreement on the analyses to be performed, assigning priority to the issue among the other workload of the agency, making a well-informed decision, and finally write a clear decision and its basis and publish the decision when needed.

2.3.1. *Using Risk Information in Regulatory Decisions*

Most of the safety regulations were established before the development of probabilistic safety analysis methods. These regulations were developed using engineering judgment and analyses to specify rules about design features, operations, and quality assurance. This deterministic approach, using conservative assumptions in analyses and supplemented by following the defense in depth safety philosophy, has generally resulted in substantial safety margins that have served the interests of safety well over the years.

To some extent, safety regulations have always been risk informed, in the sense that there was an attempt from the earliest times to design a plant's safety systems and accident mitigation systems with capabilities commensurate with the risk significance of design basis accidents thought to pose the most risk to public health and safety. These qualitative risk insights were sometimes augmented by quantitative risk analyses.

Since the introduction of a complete probabilistic safety assessment (PSA) framework in 1975, PSA methodology has matured and found widespread usage in different countries. By now, there is a vast literature on the technology and uses of PSA, and it is generally accepted among different regulatory bodies that PSA methods can be used to augment the traditional deterministic methods for regulatory decision making. In many cases, PSA provides deeper insights and a more balanced picture of the actual risks posed by operation of nuclear plants than the largely conservative deterministic analyses. At the same time, it is recognized that a PSA, like all other methodologies, has limitations in portraying the total risk at a plant. For example, a PSA cannot model safety culture and is, therefore, unable to quantify the risk impact of a poor safety culture at the plant. For this reason, regulators are generally cautious in using PSA bottom line estimates of risk (such as core damage frequency) as the sole basis for making regulatory safety decisions for a plant. However, a PSA does not have to be perfect to be of value to the regulator and the operator. Furthermore, many of the concepts in PSA need modification when the risks of waste disposal are considered, since the regulatory concepts for disposal require projecting doses many generations into the future.[85] Therefore, recognizing the strengths and weaknesses of probabilistic safety analyses, the regulator is faced with the question of how extensively to use risk information in its regulatory decision-making process.

In some countries, the regulatory body has the explicit policy to use PSA wherever practical in its decision-making process as a complement to deterministic approaches. Other regulatory bodies rely largely on deterministic regulations and methods, with only a limited use of PSA information. Some of the areas where it is generally agreed that PSA can be most useful in identifying plant vulnerabilities, ranking accident sequences according to their relative contribution to risk, ranking relative risk importance of different systems, components, and operator actions, specifying equipment allowed outage times and surveillance intervals, scheduling maintenance and outage activities, and analyzing operating events for lessons learned.

In the final analysis, there is no single approach to use risk information in decision making that is correct for all regulatory bodies. Each regulator must judge for itself how much weight should be given to risk information and at what pace to introduce risk-informed judgments into its decision-making process.

2.3.2. *Importance Measure*

The quantitative data that can be calculated from a PSA to characterize a particular phenomenon or problem, which relates to the risk or safety of a nuclear power plant, could be divided into global measure and importance measure. Global factors are used to characterize risk directly, while importance measures are used to characterize the contribution to the risk of the different basic events modeled in a PSA or the role they play in the defenses against those risks.

2.3.2.1. Global factors

A general definition of risk (or quantity or risk parameter) metrics is given that can be assessed at different levels of detail in the power plant, including the global, overall level integrating the effects of accident sequences on the public and the environment. Four successive levels of details are defined[86]:

1. Systems and components levels: including the unavailability of components, systems, safety functions, and frequency of preventable maintenance-related failures.
2. Plant level: including frequency of emergency shutdowns, significant events (i.e. core damage precursors), and core damage frequency (CDF).
3. External release: including frequency of containment failure or conditional probability of failure and large early release frequency (LERF).
4. Damage to public and social domains: including risk of individual deaths and total dose received by the population.

The global factors mainly used in applications are as core damage frequency, large early release frequency, and conditional core damage probability.

2.3.2.2. Importance factors

These factors (or importance measures) most referred to are the three conventional factors: risk reduction worth (RRW); absolute or relative risk achievement worth (RAWa or RAW), which is characteristic of the contribution to defense-in-depth; and absolute or relative Fussell–Vesely factor (FVa, FV) and Brinbaum importance (BI). Table 3 lists these measures and their definition.[87]

Table 3. Importance Measure and Their Definition.

Risk Importance Measure	Principle
Risk reduction (RR)	$\mathrm{R(base)} - \mathrm{R}(\mathrm{x_i} = 0)$
Fussell–Vesely (FV)	$\frac{\mathrm{RR}}{\mathrm{R(base)}}$
Risk reduction worth (RRW)	$\frac{\mathrm{R(base)}}{\mathrm{R}(x_\mathrm{i}=0)}$
Risk achievement (RA)	$\mathrm{R}(x_\mathrm{i} = 1 - \mathrm{R(base)}$
Risk achievement worth (RAW)	$\frac{\mathrm{R}(x_\mathrm{i}=1)}{\mathrm{R(base)}}$
Criticality importance (CR)	$\frac{\mathrm{RA} \times x_\mathrm{i}\mathrm{(base)}}{\mathrm{R(base)}}$
Partial derivative (PD)	$\frac{\mathrm{R}(x_\mathrm{i}+\partial x_\mathrm{i})-\mathrm{R}(x_\mathrm{i})}{\partial x_\mathrm{i}}$
Brinbaum importance (BI)	$\mathrm{R}(x_\mathrm{i} = 1) - \mathrm{R}(x_\mathrm{i} = 0)$

Van der Borst[87] concluded that from maintenance and operation optimization applications, a combination of RAW and FV importance should be used. In applications where ranking is important, only one importance measure is needed. To identify potential components to improve the safety, the Fussell–Vesely factor alone could be used. To identify the potential components for T&M relaxation, Brinbaum importance alone could be used.

2.4. *Technical Methods to Control Radioactive Contamination*

2.4.1. *Remediation of Contaminated Sites*

The nature of radioactive contaminants makes them unique when compared to other contaminants. First, their radioactivity (i.e. radiation hazards) means that they may be considered a risk sometimes at concentrations lower than toxic elements (e.g. arsenic or heavy metals). However, unlike these elements, radioactive contaminants are subject to radioactive decay, meaning that the impact or potential threat of the contaminant is reduced with time. The time scale over which this will occur depends on the half-life of the contaminant. Since half-lives vary from seconds to millions of years, the extent to which decay that assists remediation operations depends strictly on the contaminant(s). However, unlike organic contaminants, the decay of radioactive contaminants cannot be influenced by the remediation technology. These decay properties impact upon the efficacy of any particular remediation operation and will be central to the selection of any technology option. IAEA defined criteria to support the remediation program decision-making process. These criteria are based on the definition of six bands for possible remediation situation each cover an order of magnitude in dose or risk.[88] Band 1, including annual doses less than 10 μSv above background, represents risks that would be regarded as insignificant in the majority of situations. Criteria for risk insignificance have been determined based on the exemption and clearance level.[89] Band 2 represents annual doses in the range of tens of μSv above background; this range represents acceptable public exposure as a result of a set of planned actions, i.e., a justified practice.[88] Band 3 represents risks that might be considered reasonable as additional risks from a justified practice, provided that they were as low as reasonably achievable; this band includes annual doses in the range of hundred and some hundreds of μSv.[90,91] Band 4 represents risks corresponding to doses of the order of a few mSv/a. These doses are not considered acceptable. Band 5 (doses of tens of mSv/a) represents risks that would generally be regarded as unacceptable from any source (with the exception of necessary medical treatment). Band 6 (doses of hundreds of mSv/a or more) represents risks in terms of serious deterministic effects or a high probability of stochastic effects, which are clearly insignificant. Table 4 illustrates the IAEA criteria for the beginning of remediation program for contaminated area.[88]

Table 4. IAEA Criteria for Beginning of Remediation Program.[91]

Band	Need for Remediation Actions	Acceptability of Release	Range of Annual Doses
6	Remediation or prevent use	Not suitable for release	>100 mSv/a
5	Remediation or restrict use	Not suitable for release	10–100 mSv/a
4	Remediation decisions based on justification/optimization	Release may be possible subject to regular review of situation	1–10 mSv/a
3	Remediation unlikely unless constrained	Release possible situation may need occasional review	0.1–1 mSv/a
2	Remediation unlikely to be necessary based on radiological risks	Release likely review only if a problem becomes apparent	10–100 μSv/a
1	No remediation necessary	Can be released without controls	<10 μSv/a

A number of different nontechnical and technical factors directly impact the decision-making process of the remediation program. The nontechnical factors include economic factors, public perception/acceptance and public participation, costs, funding, and the availability of resources. The weight of these factors varies from case to case and can be driven by a variety of considerations, including technological, economic, and sociopsychological situations.

Economic and social implications that affect the remediation decisions depend on the size of the contamination problem. Those implications may occur over short- or long-time-scales. Decisions are often taken based on political grounds and not necessarily related to scientific or technical aspects of the environmental contamination problem. On the other hand, since the remediation programs are financed from public money the economic benefits, or detriments for that matter, of decisions on remediation projects need to be evaluated *a priori*.[92] Table 5 illustrates a simplified tabular method for measuring the performance and societal impact of a remediation option.[93] Environmental remediation cost is also an important factor that influences the decision of initiating a remediation program and the choice of technical option. Table 6 represents the cost estimate of a remediation program for uranium-contaminated soil and associated disposal.[94]

Technical factors include the assessment of the ability of the technology to reduce risk to the health and safety of the public and to the environment, the reliability and maintenance requirements for the technology, the infrastructure available to support the technology, the ease of accessing the technology and associated services, the risk to workers and public safety during the implementation of the technology, the environmental impacts of the technology, the ability of the technology to meet regulatory acceptance, and the obtaining of community acceptance. The

Table 5. Societal and Infrastructure Impacts.[93]

Impact Level	Societal Impacts in the Contaminated Area
1	No social or economic disruptions occur; no commercial, residential, or agricultural displacement occurs; and no adverse impacts on water resources occur.
2	In-migrating population of about 10% of the resident persons is dispersed within an area; no major social disruptions result; disruption of existing business patterns is avoided by standard economic planning measures; no adverse impacts on water resources occur, but minimal commercial, residential, or agricultural displacement results.
3	In-migrating population of approximately 10% of the resident persons is concentrated within a few communities; major upgrading of the public infrastructure is required; 25% of residents have lifestyles and values that are unlikely to match those of in-migrants; major social disruptions do not result; disruption of existing business patterns is avoided by standard economic planning measures; minor diversion of water resources from other activities occurs; half of the land is privately owned, and commercial, residential, or agricultural displacement results.
4	In-migrating population of approximately 20% of the resident population is concentrated within a few communities; major upgrading of the public infrastructure is required; affected communities have homogenous lifestyles and values that do not match those of the in-migrants; significant disruption to the existing business patterns and substantial economic decline during or after completion occurs; major diversion of area water sources occurs, resulting in impacts on development in the affected area; all land is privately owned, and commercial, residential, or agricultural displacement results.
5	Changes in the level of availability of public infrastructure include schools; police and fire services; water, sewer, and solid-waste systems; and recreation facilities.

Table 6. Remediation of Uranium-contaminated Soil and Associated Disposal Cost Estimate.[94]

	Cost Element	Unit Cost (US\$/m^3)
Conventional disposal costs	Excavation/screening	130
	Transportation	390
	Stabilization/solidification	260
	Disposal	293
	Total unit costs	1,073
Disposal costs using segmented gate system and containerized vat leaching techniques	Excavation	130
	Soil processing via SGS	78
	Well chemistry	325
	Disposal and transport	293
	Total unit costs	826

Table 7. IAEA Technology Evaluation Matrix.[95]

Evaluation Factor	Exemplary	Acceptable	Unacceptable
Performance	Near 100% removal	Removes contaminants to desired limit	Mobilized or additional contaminant
Reliability	Near 100% reliable	Available without excessive down time	Unreliable
Maintenance	Minimal	Occasional	Unavailable suppliers or at great cost
Cost	Costs recoverable against credits	Cost within acceptable levels	Excessive cost
Infrastructure support technology	Not needed or fully available and already in place	Available	Unavailable or requires significant expense to provide
Availability	Well proven	Demonstrated and available in short time frame	Unproven/early in development
Risk	No risk to public or operators	Risk to public or operators within regulatory guidelines	More risk than if nothing done
Impact on environment	Clean and green	Little effect on overall ecosystem	Significant pollution/damage
Regulatory acceptance	Exceeds regulatory standards	Meets regulatory standards	Fails regulatory standards
Community acceptance	Wholehearted acceptance without reservation	Acceptance with two-way dialog	Unacceptable

above-mentioned technical factors should be integrated in a structured approach to assess the decision-making process to select appropriate technologies. Table 7 lists a technology evaluation matrix developed by the IAEA; this matrix provides a subjective ranking scale for each factor. The scale offers three categories, which can be classified as exemplary, acceptable, and unacceptable. A technology assigned as unacceptable would be disqualified from further consideration.[95]

There are many ways to classify remediation technologies, but generally they could be classified according to their application either *in-situ* or *ex-situ*. *In-situ* remediation takes place within the soil/rock/water media, whereas *ex-situ* techniques rely on the removal of the contaminated materials (e.g. groundwater) prior to treatment. Both of these categories can be further subdivided into physical, chemical, biological, and thermal techniques. The applicability of these options to the remediation of radioactive contaminated sites are presented in Table 8 (a and b).[96,97] To support the remediation decision-making process, IAEA proposed a phased approach

Table 8(a). The Applicability of Different *In-situ* Remediation Technology to the Radioactive Contaminated Sites.[96]

Category	*In situ* Technique	Type of Application
Physical	Electro remediation, capping, barrier, hydraulic containment	Maybe suitable either in conjunction with the techniques, and/or following detailed consideration of site-specific characteristics
	Detector-based segregation	Commonly used, well-developed technology, and effective
	Soil vapor/dual phase extraction	Not applicable
Chemical	Soil flushing by chemical leaching, and surface amendment	Maybe suitable either in conjunction with the techniques and/or following detailed consideration of site-specific characteristics
	Stabilization/solidification	Commonly used, well-developed technology, and effective
Biological	Phytoremediation and monitored natural attenuation	Maybe suitable either in conjunction with the techniques and/or following detailed consideration of site-specific characteristics
	Bioremediation	Not applicable
Thermal	Vitrification	Not applicable

Table 8(b). The Applicability of Different *Ex-situ* Remediation Technology to the Radioactive Contaminated Sites.[96]

Category	*In situ* Technique	Type of Application
Physical	Electroremediation	Maybe suitable either in conjunction with the techniques and/or following detailed consideration of site-specific characteristics
	Excavation and disposal	Commonly used, well-developed technology, and effective
	Soil washing	Experimental or pilot scale
	Soil vapor extraction	Not applicable
Chemical	Soil washing by chemical treatment	Experimental or pilot scale
	Chemical treatment, solidification/stabilization, and surface amendment	Maybe suitable either in conjunction with the techniques and/or following detailed consideration of site specific characteristics
Biological	Bioremediation	Not applicable
Thermal	Vitrification	Experimental or pilot scale
	Incineration/thermal desorption	Not applicable

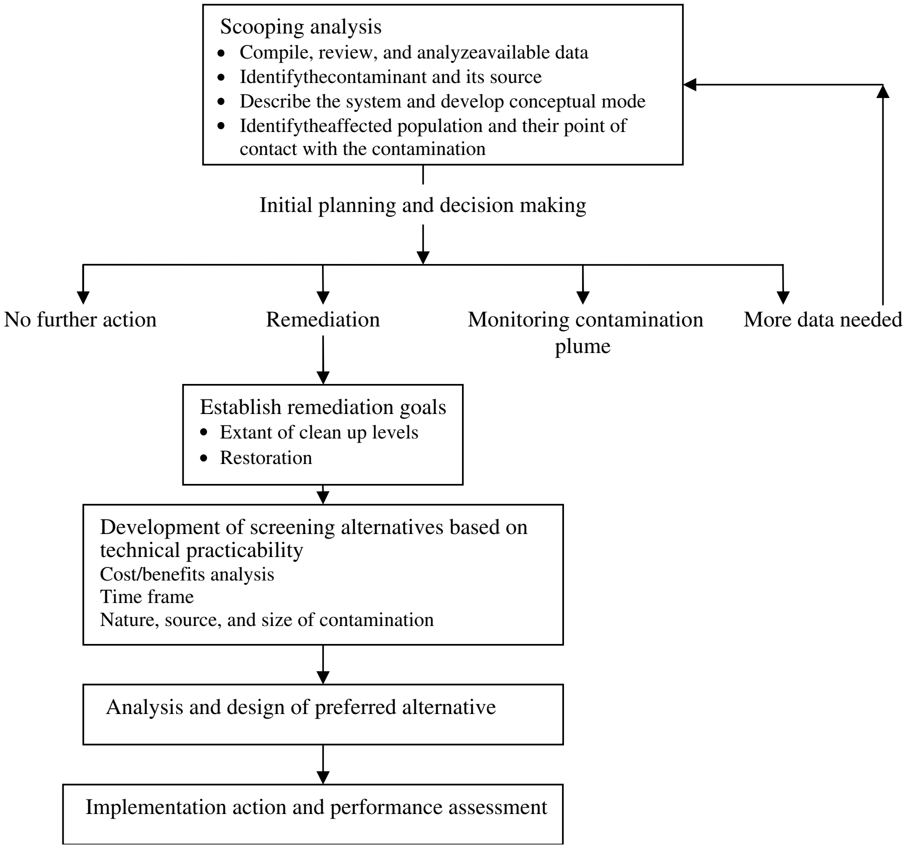


Figure 10. Phase Procedure for Support the Decision Maker.

(Fig. 10).[98] This allows for a cost-effective and governmentally sound disposition of contaminated sites.

2.4.2. *Decontamination*

Decontamination is defined as the removal of contamination from the surfaces of facilities or equipment by washing, heating, chemical or electrochemical action, mechanical cleaning, or alternative techniques.[98–100] The selection of a proper decontamination technique is not a straightforward process; it must be based on several selection criteria that include decontamination effectiveness, and impact of the decontamination technique and its associated constrains.[101]

The requirements for effective decontamination include loose debris removal, adherent particle removal, particle removal from crevices, effect of internal components, production rate, remote operation, and degree of development.[102]

- Removal of loose debris includes the removal of both the loose particles, which have deposited at the bottom portions of the contaminated system in low spots and dead volume spaces, and the loosely adherent and smearable materials that may be attached to the internal surfaces of the system.
- Removal of adherent particles includes the radioactive material that adheres tightly to a surface; more tightly than the smearable film discussed in the first requirement. It also includes corrosion layers if that layer contains radioactive material.
- Particle removal from crevices, evaluate the removal of radioactive particles that migrated into crevices such as valves, demineralizers, pumps, and tanks with internal parts can contain narrow spaces, such as cracks or crevices.
- Effect of internal components used to determine the effect of the internal components in the contaminated equipments on the applicability and effectiveness of a particular decontamination technique.
- Production rate at which a particular method will effectively decontaminate a component is an important requirement in the determination of the effectiveness of the technology, because it is significant to both cost and radiation exposure. However, since production rate is not as important as the effective removal of contaminated material, the weighting factor given to production rate is not usually so high than that assigned to the other requirements.
- Remote operation; ALARA requirement for minimum exposure to operating personnel is one of the more important considerations in selecting a decontamination method. This requirement can greatly reduce the applicability of some otherwise promising decontamination method. The need to protect personnel from excessive exposure will increase the difficulty of the cleaning operation, add to its cost, and increase the time required. The extent of these difficulties will vary inversely with the degree of remoteness of operation of a given decontamination method.
- Degree of development is defined as the extent that the decontamination technique has been developed for industrial use. The technique may have been developed to the point where it is (1) used routinely and the equipment is essentially "off the shelf" in availability. (2) proven, but is still in the developmental stage, and has been employed only on an experimental basis, or (3) still in the experimental stage, and the equipment required for its application requires development rather than refinement.

In addition to the requirements relating to the effectiveness of the different decontamination techniques, there are other requirements concerned with impacts and constraints associated with use of the techniques that include[102]:

- Radiological safety aspects of a particular technique that can greatly influence its latitude for use.

- Secondary waste generation that must be reduced and properly packaged for storage, transport, and disposal. The radioactive waste that is removed from the contaminated surfaces can be expanded in volume by the decontamination agent during the decontamination process and this will lead to increase the disposal cost.
- Need for disassembly, some of the contaminated equipments would have to be disassembled to some degree to provide access to the interior surfaces. Both the degree of disassembly required and the relative difficulty of disassembly would impact the applicability of a particular decontamination technique.
- Accessibility of the contaminated system component is a very important requirement. Depending upon the decontamination technique, one or more openings may be required to decontaminate the system at hand.
- Size of item: for off-system decontamination, large items might be unsuitable to fit off-system decontamination equipment, e.g., an ultrasonic tank. These items must be sectioned.
- Capital cost: initial cost of the decontamination equipment and supporting systems and the expected life of equipment are important factors in the selection of an appropriate decontamination method.
- Operating cost includes the labor and consumable supplies costs. Labor intensive decontamination technique, and/or one that requires a large amount of expensive material is unfavorable. It should be noted that the operating cost of the radioactive contaminated material can be quite different from the cost of nonradioactive industrial cleaning and this is related to the additional costs associated with personnel shielding and waste disposal.
- The feasibility of recycling the contaminated component or system after decontamination can be affected by a number of different factors, which includes the comparison between the decontamination cost the replacement cost. Another factor is the effect of access or sectioning operations that require excessive repair work. In addition, the amount of the removed metal from the decontaminated surface and the final surface quality, since rough surfaces are much more subject to rapid recontamination.
- The tendency to corrode the surface as a result of the reaction between the decontamination agent and the item being decontaminated, both during the decontamination and as a residual effect later on.
- Inherent safety characteristics of the decontamination technique.

Chemical decontamination is usually carried out by circulating chemical reagents in the contaminated system; they are generally most effective on nonporous surfaces. The choice of decontamination agents is based upon the chemistry of the contaminant, the chemistry of the substrate, and the ability to manage the waste generated during the process. The mechanical decontamination is used on any surface where contamination is limited to near surface material. Alternative decontamination techniques include electropolishing, at which an anodic dissolution technique is used where a controlled amount of material is stripped from the contaminated surface along with the contamination. The process works for any conductive metal, providing protective surface coatings are not present, but the choice of electrolyte is important. The components are decontaminated following removal by immersing them in a bath of fluid. Table 9 lists different decontamination techniques and their

Table 9. Applicability of Decontamination Techniques for Different Materials and Surfaces.

Technique	Material	Application Material
Chemical	Strong mineral acids	Stainless steel, inconel, carbon steel, metals, and metallic oxides
	Acid salts	Metal surfaces
	Organic acid	Metal and plastic surfaces, stainless steel, and carbon steel
	Bases and alkaline salts	Carbon steel
	Complexing agents	Metals
	Bleaching	Organic material from metals
	Detergents, surfactants, and organic solvents	Organic materials from metals, plastics, and concrete
	Foam and chemical gels	Porous and nonporous surfaces
Mechanical,	Flushing with water	Large areas (too large for wiping or scrubbing)
	Dusting/vacuuming wiping/scrubbing	Concrete and other surfaces
	Strippable coating	Large nonporous surfaces and easily accessible
	Steam cleaning	Complex shapes and large surfaces
	Abrasive cleaning	Metal and concrete surfaces and hand tools
	Sponge blasting	Paints, protective coatings, rust, and metal surfaces
	CO_2 blasting	Plastics, ceramics, composites, stainless steel, carbon steel, concrete, and paints
	High-pressure liquid nitrogen blasting	Metals and concrete
	High-pressure and ultra-high-pressure water jets	Inaccessible surfaces structural steel and cell interiors
	Grinding/shaving	Floors and walls
	Scarifying/scabbling	Concrete and steel surfaces planning
	Milling	Large number of similarly shaped items
	Drilling and spalling	Concrete only
	Expansive grout	Thick layers of contaminated concrete
Alternative	Electropolishing	Conductive surfaces
	Ultrasonic cleaning	Small objects with loosely adhering contamination
	Melting	Metal

applications.[103] An overview of the strength, limitation, and costs of different decontamination technologies is shown in Table 10.[104]

3. Control of Radioactive Waste

3.1. *Sources of Radioactive Wastes*

3.1.1. *Types of Radioactive Wastes*

The types of the generated radioactive wastes can be divided into aqueous waste, liquid organic waste, solid waste, wet solid waste, biological waste, and medical waste.[106]

3.1.1.1. Aqueous waste

Aqueous waste is generated during nuclear reactor operations and in other operations involving the application of radioisotopes (e.g. medicine, research, and education). The type of liquid waste produced depends upon the particular operation being conducted and can vary extensively in both chemical and radionuclide contents. Most operations, particularly the larger ones, also produce a variety of radioactive liquid wastes from locations such as showers, laundries, and analytical laboratories and from decontamination services. The specific activity of the waste generated depends upon which radioactive materials are used.

3.1.1.2. Liquid organic waste

Liquid organic waste is generated from medical, industrial, and research centers forms a relatively small volume compared with other radioactive wastes. Typically, this waste includes oils, solvents, scintillation fluids, and miscellaneous biological fluids.

3.1.1.2.1. *Oils*

Radioactive oil waste consists of lubricating oils, hydraulic fluids, and vacuum pump oils. This type of waste generally contains only relatively small quantities of beta/gamma-emitting radionuclides, but may also contain trace quantities of alpha-emitting radionuclides, depending on its origin. This waste generally arises from activities in nuclear research centers; tritium contaminated oils may also arise from various medical and industrial applications. Radioactivity levels for oils may vary widely, depending on the applications they are associated with.

Table 10. Overview of Different Decontamination Technology.[104]

Technology	Strengths	Limitations	Special Considerations	Cost (US$/$m^2$)
Organic acids	Applied to wide range of contaminants. Safer than other chemical techniques.	Requires considerable on-hand chemical knowledge for best application.	Contaminant solubilization requires great care in waste treatment. Danger of mobilization of the contaminant.	10.76
Strong mineral acids	Remove very stubborn deposits. Much operating experience from industrial cleaning.	Great care needed operationally due to safety considerations, as it can destroy substrate.	Primarily used for metal corrosion products.	21.53
Chemical foams and gels.	Increased contact time aids performance. Can reach remote and hidden areas.	May require repeated applications to achieve maximum effectiveness.	Care must be taken when flushing since foams can travel to the areas beyond the reach of liquids.	21.53
Oxidizing and reducing agents.	Disrupts matrix where contaminants hide so small amounts can be very effective.	Must be targeted at appropriate situation. Will not work if redox chemistry is not suitable.	Often used as one step of a multiple-step process.	21.53
Strippable coatings	Produce single solid waste. No airborne contamination and no secondary liquid waste.	The spray gun nozzles clog. From a cost perspective, may be best suited for smaller decontamination activities.	Works for only easily removed (smearable) contaminants.	52.20
Centrifugal shot blasting	Good at removing paint and light coatings from concrete surfaces in open areas.	Escaped shot may pose a hazard to workers. Require air compressor, systems for dust collection, and air filtration.	Can be limited by large size, hence unable to get into corners.	368.66
Concrete grinder	Fast and mobile. Less vibration.	Small size limits utility.	Used in combination with other technologies.	31.43

(*Continued*)

Table 10. (*Continued*)

Technology	Strengths	Limitations	Special Considerations	Cost (US$/m^2)
Concrete shaver	Good for large, flat, and open concrete floors and slabs. Fast and efficient.	Does not maneuver well over obstacles. Good for only concrete floors and slabs.	Attractive alternative to handheld scabblers.	14.21
Concrete spaller	Good for in-depth contamination. Fast.	Requires predrilling of holes. Leaves behind a rough and uneven surface.	Limited commercial availability.	199.35
Dry ice blasting	Very good for contamination on a surface.	Cannot remove deep contamination.	Requires support systems: air compressors, dryers, and filters.	N/A
Dry vacuum cleaning	Readily available. Works well with other physical decontamination technologies.	Good for only loose particles.	Typically used in conjunction with other decontamination technologies.	21.53
Electrohydraulic scabbling	Generates less secondary waste than other technologies using water. Very efficient. Removes deep contamination.	Requires a skilled operator. Generates some secondary liquid wastes.	Works best for horizontal surfaces.	107.64
En-vac robotic wall scabbler	Works on large open spaces, including walls and ceilings. Worker exposure to contaminants is limited: remote operation.	Requires additional attachments to address irregular surfaces, obstacles, and tight.	Remote-controlled aspect allows operation in areas unsafe for humans.	139.35
Grit blasting	Different types of grit and blasting equipment are available for a variety of applications.	Generates large amounts of dust and particulates during operation.	Wide range of grits and abrasives are available for special situations.	Based on En-vac system.
High-pressure water	High-pressure systems are readily available.	Generates a significant secondary waste stream.	Can physically destroy substrate. Best used on sturdy structures.	39.07

(*Continued*)

Table 10. (*Continued*)

Technology	Strengths	Limitations	Special Considerations	Cost (US\$/m^2)
Soft media blast cleaning	Removes virtually all of the contamination from the surface.	Generates significant amounts of airborne contamination. Lower productivity.	Applicable to surface decontamination only.	49.51
Steam vacuum cleaning	Easy. Washed surfaces dry quickly. Good for large flat surfaces.	Not good for grease. Poor ergonomic design. Not good for irregular surfaces.	Not recommended for surfaces that can be damaged by steam temperatures.	146.82
Piston scabbler	Remotely operated and available. Good for open, flat, and concrete floors and slabs.	Units are loud. Remote units cannot operate close to wallfloor interfaces.	Remote-controlled aspect allows operation in the areas that are unsafe for humans.	64.58

3.1.1.2.2. *Scintillation liquids*

Scintillation liquids result from radiochemical analyses of low-energy beta emitters, such as ^{3}H and ^{14}C. They typically consist of nonpolar organic solvents such as toluene, xylene, and hexane; however, they may also include biological compounds such as steroids and lipids.

3.1.1.2.3. *Solvents*

Spent solvents may arise from solvent extraction processes. The most commonly used extraction solvent is tributyl phosphate (TBP). TBP is diluted for the extraction process usually with a light saturated hydrocarbon, often dodecane or a mixture of paraffin. A variety of organic decontamination liquids and solvents, such as toluene, carbon tetrachloride, acetone, alcohols, and trichloroethane, arise from various operations.

3.1.1.3. Solid waste

Solid waste can be segregated into two main groups; which are compactable and combustible solid waste and noncompactable and noncombustible solid waste.[107] The largest volume of solid waste is general rubbish, which includes protective clothing, plastic sheets and bags, rubber gloves, mats, shoe covers, paper wipes, rags, towels, metal, and glass.

3.1.1.4. Wet solid waste

Wet solid waste, such as spent radioactive ion exchange resins, precipitation sludges, and evaporator concentrates, is generated by the treatment of aqueous waste streams at nuclear research centers or at centralized waste processing facilities.

3.1.1.4.1. *Spent ion exchange resins*

Ion exchange media can be classified into two basic categories: inorganic ion exchangers (both natural and synthetic) and organic resins (mainly synthetic). Most commercial ion exchangers are synthetic organic resins typically consisting of polystyrene cross-linked with divinylbenzene. Spent organic and inorganic ion exchange media may require different treatment and conditioning options. Although regeneration of spent organic resin is possible, the preferred option is direct conditioning of spent resin, as regeneration results in the production of highly acidic and caustic radioactive liquids, which may be difficult to treat.[108]

3.1.1.4.2. *Precipitation sludge*

The product of treatment of liquid radioactive waste by chemical precipitation and flocculation is a sludge containing most of the radioactivity; this can vary greatly in terms of its chemical and physical characteristics, depending on the specific process used.[109] The chemical composition of the sludge differs from the initial waste owing to the addition of the precipitating chemicals.

3.1.1.4.3. *Evaporator concentrates*

Evaporator concentrates are produced through an evaporation process by which the volatile and nonvolatile components of a solution or slurry are separated to reduce both the waste volume and the amount of radioactivity in a liquid effluent. Evaporation is most effectively used for radioactive liquids with high concentrations of salts or other impurities. The concentrate or bottoms product can range from 15 wt% solids to a virtually dry powder or cake, depending on the evaporator type and efficiency and on the chemical composition of the waste stream.[110,111]

3.1.1.5. Biological waste

Biological waste arises from biological, research, and teaching/training practices. This waste includes animal carcasses, contaminated body fluids, and animal tissues. The inclusion of materials having a biological origin clearly distinguishes this type of waste from inorganic materials. A primary example of biological waste is the waste from research involving animals. All discharges (e.g. feces, urine, and saliva) from animals used in research involving radioactive materials must be considered to be potentially contaminated. Animal cage containers must be treated as contaminated until monitored and declared free from contamination.

3.1.1.6. Medical waste

Medical waste may be defined as radioactive waste arising from diagnostic, therapeutic, and research applications in medicine. In addition to being contaminated by radioactivity, medical waste, such as biological waste, can have infectious, pathological, and other hazardous properties. In many instances, the potential additional hazard, either from the waste's chemical, biological, or physical properties, is greater than the radiological hazard.[112] The following types of radioactive waste may occur because of the use of radionuclides in medicine:

- Spent radionuclide generators and spent sealed radiation sources;
- Anatomical and biological wastes (e.g. body parts, tissues, organs, fluids, and excreta from patients administered with radionuclides);
- Miscellaneous aqueous and organic liquids and radioactive solutions;
- Miscellaneous solid dry waste (e.g. gloves, paper tissues, and equipment parts); and
- Miscellaneous waste posing a puncture hazard (e.g. needles, broken glass, and nails).

3.2. *Principles and Objective of Managing Radioactive Wastes*

The International Atomic Energy Agency (IAEA) has developed a comprehensive set of principles for the safe management of radioactive wastes. These principles are applicable to all countries and can be applied to all types of radioactive wastes, regardless of its physical and chemical characteristics or origin. They include the protection of human health and the environment, now and in the future without imposing undue burden now or in the future.[113] In addition to the internationally accepted principles, each country has its own policies that define the aims and objective for the regulatory framework; these might includes administrative and operational measures (i.e., control of radioactive waste generation, safety of facilities, and decision-making criteria).[114]

One of the most important radioactive waste management principles is the control of radioactive waste generation. The objective of this principle is to limit the generation and spread of radioactive contamination and to reduce the volume of wastes for storage and disposal, thereby limiting any consequent environmental impact, as well as contaminated material management costs.[103] The main elements of a waste minimization strategy can be grouped into the following four areas:

1. Keep the generation of radioactive waste to the minimum possible or practicable in terms of both its activity and its volume, by appropriate design measures, facility operation, and decommissioning practices. This includes the selection of appropriate technology, the selection and control of construction and operational materials, the recycle and reuse of materials, and the implementation of appropriate procedures.
2. Minimize the spread of radioactivity leading to the creation of radioactive waste as much as possible by containing it to the possible greatest extent.
3. Optimize possibilities for recycle and reuse of valuable components from existing and potential waste streams. Implementation of recycle and reuse options requires the availability of suitable criteria, measurement methodology, and instrumentation. IAEA proposed a linear decision-tree approach, which could be adopted to evaluate the factors that influence a specific recycle and reuse option, as indicated in Fig. 11.[103]
4. Minimize the amount of radioactive waste that has been created by applying adequate treatment technology.

There are different factors that influence the waste minimization principles such as the technical feasibility and availability of the technology for waste minimization, economic consideration, radiological factors applied to release practice, hazards and risks, national policy, regulatory climate, public acceptance, and legal liability.

International surveys indicate that the applied release practice criteria vary widely among various countries. These criteria might be based on nationally applicable regulations or on a case by case evaluation. Historical examples of clearance criteria from specific projects in various countries are indicated in Table 11.[103] The national regulation in some countries set limits for alpha emitters as three to ten times higher for smaller contaminated areas (hot spots). In some countries, the limits for alpha and beta-gamma are specified separately, while others maintain a specified limits (France, Germany, Sweden, United Kingdom, and the United States). Some of the regulations specifically indicate that decontamination prior to clearance is considered acceptable (Belgium, Germany, and the United States). In Germany, a specific formula has also been applied in some projects/plants to set limit values for those nuclides that can be handled without regulatory control. In addition, further restrictions have been applied in terms of total activity, total mass, and total volume in some of the projects/plants in Belgium, Germany, and Sweden.

3.3. *Elements of Radioactive Waste Management System*

During the establishment of a waste management system, all stages in waste processing are considered, starting from waste generation through sorting and treatment

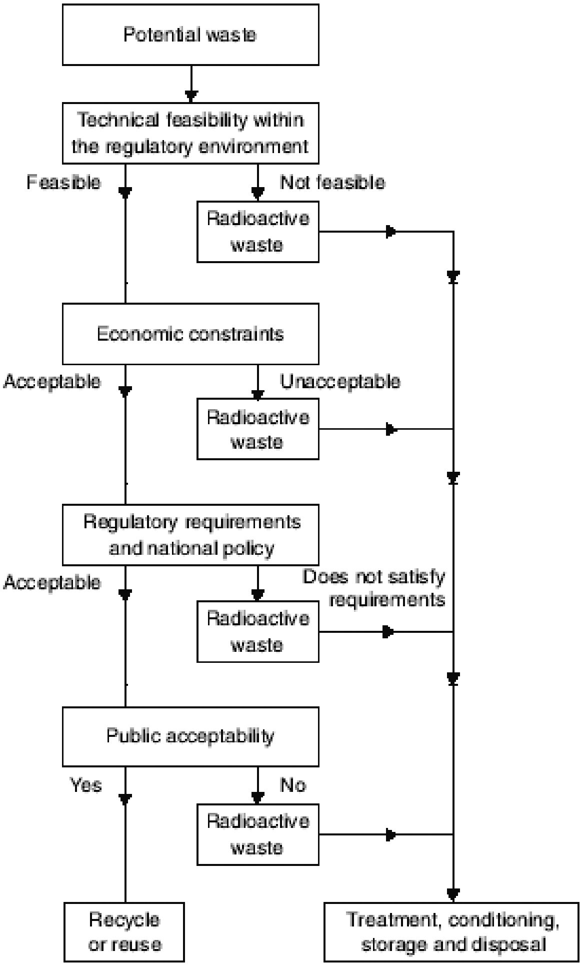


Figure 11. Linear Decision Tree Approach for Recycle and Reuse Applications.[103]

until disposal of these wastes. The basic steps of the radioactive waste management system are depicted in Fig. 12.[115] To achieve the overall goal of safety in waste management, component elements should be complementary and compatible with each other. The core of the waste management system is the technologies that are applied to the waste from its generation to its disposal.

3.3.1. *Transportation*

Safety in the transport of radioactive material is provided through meeting the provisions of transport regulations, which aim to protect persons, property, and the environment from the effects of radiation during the transport of these materials. The transport regulations include requirements on the waste package that ensure

Table 11. Examples of Surface Contamination Limits for Beta–Gamma Emitters Applied in Specific Projects for Unrestricted Reuse or Unrestricted Disposal.[103]

Country	Surface Contamination (Bq/cm^2)	Additional Information Limit
Germany	0.37	Averaged over 100 cm^2 for fixed and removable contaminations and for each single item. Applied to scrap metal originating from nuclear installations.
	0.50	Applied to scrap metal and concrete originating from nuclear installations.
Slovakia	0.37	Case-by-case decision on materials from decommissioning, 100% direct surface measurements.
Finland	0.40	Removable surface contamination over 0.1 m^2 for accessible surfaces. Applied to radioactive substances originating from application in nuclear energy production.
Belgium	0.40	Mean value for removable surface contamination over 300 cm^2, for beta–gamma emitters and alpha emitters with low radiotoxicity.
The USA	0.83	Surface contamination above background over no more than 1 m^2, with a maximum of 2.5 Bq/cm^2 above background if the contaminated area does not exceed 100 m^2.
Italy	1.00	Case-by-case decision for a limited amount of material from decommissioning.
Canada	1.00	Averaged over 100 cm^2 for total contamination, 100% survey of all surfaces.
France	3.70	Materials from decommissioning, 100% direct surface measurements.
Sweden	4.00	Mean value for removable surface contamination over 100 m^2, with a maximum of 40 Bq/cm^2 if the contaminated area does not exceed 10 cm^2. Applied to radioactive substances originating from application in nuclear energy production.
India	4.00	Averaged over 100 cm^2 for fixed uranium contamination. Applied to scrap metal originating from refining facilities. The material is considered for free release if the concentration of uranium in the slag is less than 4 ppm.

its survival under accident conditions. Transport plans can be of a general nature or can be submitted on a case-by-case basis. Simplified transport plans may be used for the transport of relatively small quantities of material, as illustrated in Fig. 13.[116]

3.3.2. *Treatment*

Treatment refers to operations that reduce the volume of the generated wastes. There are various volume reduction technologies[116]; the selection of any of these technologies is largely depending on the waste type. The radioactive waste can be

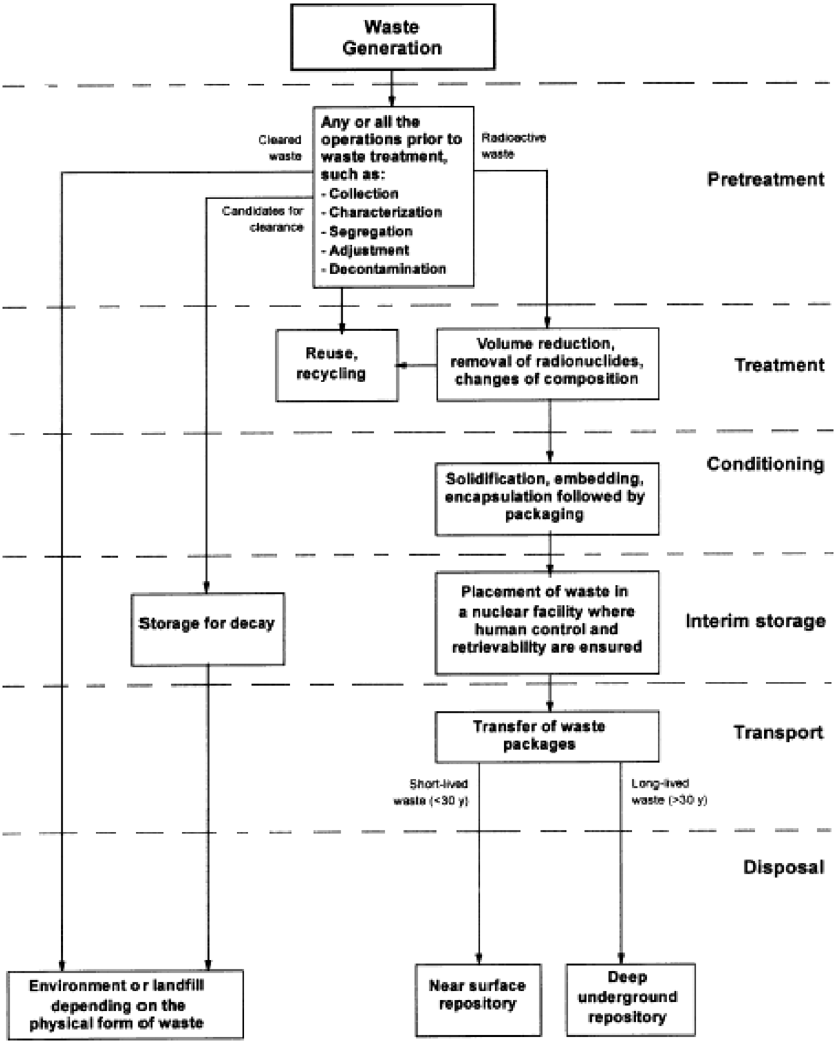


Figure 12. Waste Management Steps.

classified according to its chemical and physical characteristics. Tables 12 and 13 present a general guide showing the main features and limitations of some available aqueous and organic liquid radioactive waste treatment processes.[117] Figure 14 illustrates different decontamination ranges for different treatment options for aqueous liquid radioactive wastes.[118]

3.3.3. *Conditioning*

Conditioning includes operations that produce a waste package suitable for handling, transportation, storage, and disposal.[118,119] The immobilization of radioactive waste

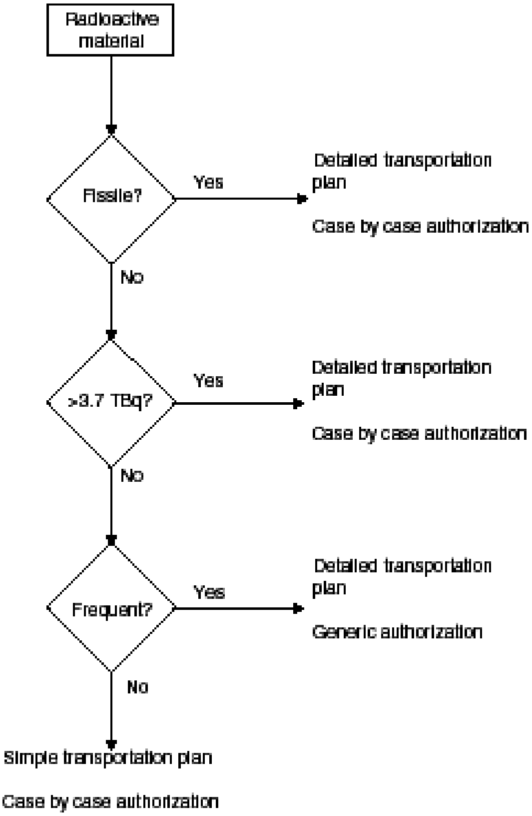


Figure 13. Transport Authorizations Options.[115]

to obtain a stable waste form is an important step to minimize the potential for the migration or dispersion of radionuclides into the environment during storage, handling, transportation, and disposal. When the immobilized waste is packaged, the resulting waste packages must be capable of meeting shielding and containment requirements for handling, storage, transportation, and finally the waste disposal site requirements. The choice of the immobilization waste matrix depends on the physical and chemical nature of the waste and the acceptance criteria for the disposal facility to which the waste will be disposed. Table 14 compares different waste immobilization matrices used in the conditioning of low-level and intermediate-level radioactive wastes.[117]

3.3.4. *Storage*

Storage is an integral part of the waste management system. The main functions of a storage facility are to provide safe custody of the waste packages and to protect both operators and the public from any radiological hazards associated with radioactive

Table 12. Main Features of the Aqueous Waste Treatment Processes.[115]

Treatment Method	Features	Limitations
Chemical precipitation (coagulation/ flocculation/ separation)	Suitable for large volumes and high salt content waste. Easy industrial operations. Not expensive.	Generally lower DF than other processes ($10 < \mathrm{DF} < 10^2$). Efficiency depends on solid–liquid separation step.
Organic ion exchange	DF good on low salt content. Good mechanical strength. Regenerable.	Limited radiation and thermal and chemical stabilities. Resins cost important. Immobilization difficulty.
Inorganic ion exchange	Chemical, thermal, and radiation stabilities better than organic ion exchangers. Relatively easy immobilization. Large choice of products ensuring high selectivity $\mathrm{DF} > 10–10^4$.	Affected by high salt content. Blockage problems. Possible high cost. Regeneration and recycling often difficult.
Evaporation	$\mathrm{DF} > 10^4–10^6$. Well-established technology. High volume reduction factor. Suitable for a large number of radionuclides.	Process limitations (scaling, foaming, corrosion, and volatility of certain radionuclides). High operation and capital costs.
Reverse osmosis	Removes dissolved salts. DF $10^2–10^3$. Economical. Established for large scale operations.	High pressure system, limited by osmotic pressure. Non-backwashable, subject to fouling.
Ultrafiltration	Separation of dissolved salts from particulate and colloidal materials. Good chemical and radiation stabilities for inorganic membranes. Pressure <1 MPa.	Fouling-need for chemical cleaning and backflushing. Organic membranes subject to radiation damage.
Microfiltration	Low pressure operation (100–150 kPa). High recovery (99%). Excellent pretreatment stage. Low fouling when air backwash Employed.	Backwash frequency can be high; depends on solid content of waste stream.
Electrochemical	Low energy consumption. Enhances the effectiveness of reactions.	Sensitive to impurities in waste stream. Ionic strength of waste stream can effect performance. Fouling is a problem above 10 g/L total solids.

(*Continued*)

Table 12. (*Continued*)

Treatment Method	Features	Limitations
Solvent extraction	Selectivity enables removal, recovery, or recycle of actinides.	Organic material present in aqueous raffinate. Generates aqueous and organic secondary wastes.

Table 13. Main Features of Liquid Organic Waste Treatment Methods.[115]

Treatment Method	Features	Limitations
Incineration	Decomposes organic nature of waste. High volume reduction. Combined use for other waste. Eliminates infectious hazard.	Secondary waste must be treated. High temperatures are required to ensure complete decomposition. Off-gas filtration and monitoring are required.
Emulsification	Allows embedding of liquid organic waste into cement matrixes.	Low limitations for content of emulsified liquids in the cement matrix.
Absorption	Solidifies and immobilizes organic liquids. Simple and cheap.	Suitable only for small amounts of waste. Absorbed waste may not meet disposal acceptance criteria.
Phase separation (e.g. distillation)	Produce clean solvent. Removes water and detoxifies the waste for direct disposal.	Nonuniversal application. Technology is relatively expensive for this type of waste.
Wet oxidation	Low temperature process. Simpler than incineration. Suitable for biological waste.	Requires storage of oxidizing agent. Residue requires immobilization.

waste. The design of storage facilities must meet national regulatory standards and basic safety principles. Storage facilities are designed to facilitate inspection and monitoring of stored waste, keep exposure to personnel as low as reasonably achievable principle (ALARA), and provide adequate environmental conditions to ensure proper conservation of waste packages during their tenure at the facility. A key function during storage is the maintenance of records allowing the identification of stored waste packages.

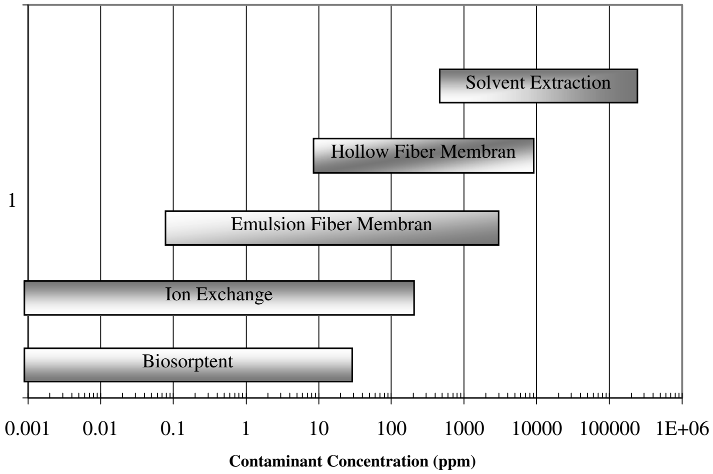


Figure 14. Decontamination Ranges for Different Treatment Technologies (Developed Form Kentish and Stevens, 2001).[116]

Table 14(a). Comparison between Different Immobilization of Process Characteristics.[117]

Process Characteristics	Cement	Polymer	Bitumen
Complexity	Low	High/average	High
Flexibility	High	Average	High
Volume reduction	Negative	Negative	Positive
Cost	Low	High	High

Table 14(b). Comparison between the Applicability of Different Immobilization of Process.[117]

Waste Form	Cement	Polymer	Bitumen
Compatibility with waste streams	Average	Average	High
Waste loading	Average	High	High
Compressive strength	High	Average–high	Low
Impact resistance	High	Average–high	Average
Fire resistance	High	Low–average	Low
Radiation stability	High	Average	Average
Retention of actinides	High	Low	Low
Retention of short-lived	Low	High	High

3.3.5. *Disposal*

The basic objective of disposal is to isolate the waste from water and the human environment under controlled conditions allowing the radioactivity either to decay naturally or to slowly disperse to an acceptable level. The choice of a disposal option depends on the waste type and on local conditions, including geological and hydrogeological conditions, radiological performance requirements, and considerations of sociopolitical acceptance.

3.4. ***Disposal of Radioactive Waste***

3.4.1. *Life Cycle of the Disposal Facility*

The life cycle of any disposal facility comprises different major phases,[121,122] as shown in Fig. 15, covering different types of activities and requiring specific strategies, political decisions, and appropriate human and technical resources. During the lifecycle of the disposal project, there is a need of identifying and assessing the potential nontechnical factors that might impact the disposal project. The greatest impacts generally occur during the construction, operation, and closure phases. The significant of each factor varies from disposal project to another. These factors includes[123]:

- Land resources: Displacement of land resources might result during the excavation of soils, aggregate for road construction, waste cover material, and disposal unit

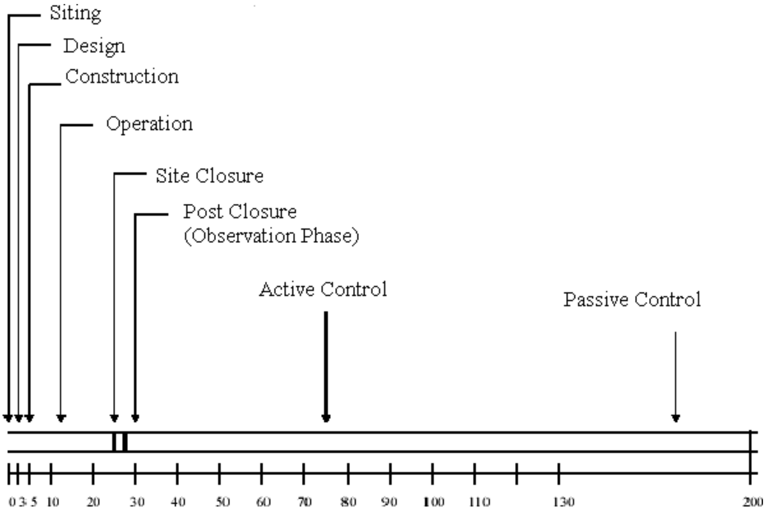


Figure 15. Phases of Disposal Life Cycle.[121]

capping. These disposal materials may be obtained from on-site or off-site sources, resulting in some impact at these sources.

- Ecologically sensitive areas: Areas identified as ecologically sensitive, such as the habitat of rare or protected plant or endangered animal species or special wetlands, may be affected by disposal development, including the potential for erosion of disturbed soils.
- Air quality: Generation and dispersion of dust from increased vehicle traffic and the emission of combustion engine may affect ambient air quality.
- Groundwater resources: Groundwater may be withdrawn to meet the project infrastructure water requirements, potentially including dust control, cement batch plant operation, waste container grouting, drinking water, or septic system or sewers. The withdrawn groundwater may affect well usage, springs, or wetlands in the vicinity of the repository. In addition, the disposal may adversely impact groundwater quality in case of contaminant leakage from the disposal facility.
- Surface water resources: Engineered storm water control features may contribute effluent to surface water bodies or drainage systems and may cause erosion.
- Biotic resources: During the project, a removal of some land area from the baseline plant (flora) and animal (fauna) habitat occurred. This may affect species present on or near the disposal site and along transport routes to the disposal. For wildlife, impacts extend to the home range of movement of the affected species, e.g. feeding and movement territories. In addition, disposal impacts on surface or groundwater resources may in turn affect flora and fauna.
- Visual impacts: The appearance of the natural landscape is likely to be changed by the development of the disposal and related infrastructure construction.
- Historic or archaeological sites: Disposal development, affecting such sites, structures, or artifacts, may alter or destroy historically or archaeologically significant resources, or impair their preservation for future use and enjoyment.
- Demographic: Depending on the size and nature of the project, increases in population may occur in the local community due to incoming workers and family members, especially if the initial size of the host community is relatively small. These changes can affect housing, community social services and infrastructure demands, and community character.
- Social structure: Changes could result if the income levels and educational background of the incoming workers varies significantly from the existing social structure in the local community.
- Community character: During siting, impacts in the local community and adjacent areas may occur based on varying opinions about the proposed disposal. The involvement of interested parties from outside the local community may increase these impacts. Community views may range from perceptions of an undesirable

image and related social tensions to support the economic development and job creation benefits.

- Community health: The nature of the facility to be built may cause anxieties and fears in some individuals and groups that may result in potential human health impacts, especially during the early phases of the disposal development process.
- Employment and labor supply: Disposal development is generally accompanied by local job creation. The total number and the skill levels required will vary depending on project size, nature of wastes accepted, and the technology utilized. New workers may be drawn from the local or surrounding community or the outlying region if the disposal is in an area remote from populated areas. Employment opportunities may be seen as a local benefit. However, the extent of the opportunity depends on the required skill sets. Trade union provisions may also apply. Other local employers may experience a decrease in available skilled workers and perhaps upward pressure on wage levels. Employment needs may fluctuate considerably during different disposal life cycle phases.
- Local economic activity: The project is accompanied by direct purchase of materials, supplies, buildings, vehicles, equipment, fuel, lodging, restaurant meals, professional, and trade services. This purchasing may represent an opportunity for local and regional suppliers and also could result indirectly in new business development. Depending on the level of direct disposal-related spending, these expenditures may have a significant multiplier effect on local and possibly regional economic development. Business development may include complementary nuclear- and engineering-related industries, such as a waste treatment facility, batch cementation plant, or a container fabrication plant.
- Housing: The influx of new employees and their families may place demands on available housing stock, both for rental and ownership, possibly resulting in higher housing costs, increased property values, a shortage of housing, and a potential need to provide additional temporary and permanent housings, depending on the size of the repository. The closure of the facility could result in a surplus of housing stock, which affects housing-market activity. Alternatively, the concern regarding the radioactive waste in the disposal could adversely affect housing-market activity and depress property values. The extent of these impacts may differ considerably during the different disposal phases.
- Education: Depending on the size of the disposal, incoming workers and their families may also put pressure on local educational facilities if there are not sufficient numbers of teachers and classrooms to accommodate new students. Where education provision is the responsibility of the local administrative body, the body may not have the resources to respond the demand for new facilities. During the closure of the disposal, the demand on the educational system will be reduced, possibly resulting in surplus facilities and staff.

- Transportation network: If the shipment of waste to the disposal is by road, this transport will increase traffic levels and possibly road maintenance needs. Construction of new access roads or the improvement of existing roads or the provision of new rail access may be required. Where the local road network is the responsibility of the local administrative body, the body may not have the resources to respond to the potential need for road upgrades or maintenance needs.
- Community services: Depending on the size and nature of the facility, disposal development may produce direct and indirect demands on local community services and facilities, especially if the initial size of the host community is relatively small. These services may include the provision of police and fire protection, hospitals and other health care facilities, social services, emergency response services, and public transportation. Funding for these services may come from a variety of sources. Where community services are the responsibility of the local administrative body, the body may not have the resources to respond to the increasing demands for some of these services. These impacts may arise during the construction, operation, and closure phases.
- Utility availability: The project requires electric power, potentially involving transmission line extension or electricity substation development. Water use and wastewater discharge may require connection to off-site infrastructure, or provision of an on-site water supply well or septic system. Where utility services are the responsibility of the local administrative body, the body may not have the resources to respond to increasing demands. These impacts typically occur during the construction and operation phases, diminish during closure, and are minimal at post-closure, e.g. some provision may be needed for ongoing storm water drainage.
- Park and recreational lands: Lands set aside for parks, hunting, hiking, fishing, or other recreational uses may be affected if disposal development would restrict or prevent future use, or impair the quality of recreational activities. Where park and recreational lands are the responsibility of the local administrative body, the body may not have the resources to respond to increasing demands. These impacts may arise during the construction, operation and, to a lesser degree, closure phases.
- Development plans: Depending on the size and nature of the facility, disposal development may affect the existing land uses and future plans. Concerns regarding the radioactive nature of the waste could adversely affect the future development opportunities. The waste facility itself may not be compatible with current development plans. The ability of the local planning authority to accommodate disposal siting and operation may depend on the size of local and regional communities and on previous experience and attitude toward similar industrial development. After the closure of the repository, the local planning authority may have alternative plans for the future use of the site. These impacts typically occur during the siting, construction, and post-closure phases.

Table 15. Potential Impacts During Disposal Life Cycle Phases.[122]

	Siting	Design	Construction	Operation	Closure	Post-closure
Natural environment						
Land resources			x	x	x	X
Ecologically sensitive areas			x	X		
Air quality			x	x	X	
Groundwater resources			x	x	x	x
Surface water resources			x	x	X	
Biotic resources			x	x	x	x
Visual landscape			x	x	x	x
Historical or archaeological sites	x	x				
Social conditions						
Demographic			x	x	X	
Social structure			x	x	X	
Community character	x	x	x	x	X	
Community health	x	x	x	X		
Economic conditions						
Employment and labor supply			x	x	X	
Local economic activity			x	x	x	
Built environment						
Housing	x		x	x	x	
Education			x	x	x	
Transportation network			x	x	x	
Community services			x	x	x	
Utility availability			x	x	x	
Land use						
Park and recreational lands			x	x	x	
Development plans	x	x			x	

Table 15 summarize these factors and their influence on the disposal project during its different phases.[123]

3.4.1.1. Preoperational phases

These phases cover the period during which a disposal concept is selected and developed in relation to site-specific conditions. The siting process involves close integration between the selection of a suitable site and the development of an adequate disposal design. In recent years, progressively greater emphasis has been placed on meeting social objectives in siting, including acceptance by the local population. These approaches have led to two notable successes in siting new disposal facilities: at Wolsong, Republic of Korea, and at Kincardine, Canada. The objective of the activities in the preoperational phases is licensing of the facility, leading to the construction of the disposal facilities. In this section, the requirements for site selection criteria, disposal design, and waste acceptance criteria will be presented.

3.4.1.1.1. *Site selection criteria*

Guidelines for the selection of a potential disposal site usually include both technical and nontechnical criteria. Typical examples include availability of favorable geology, hydrogeology, and topography; absence of natural resources; avoidance of areas of special cultural or ecological interest; and availability of local infrastructure, including utilities, human resources, transportation routes, and basic physical services.

Candidate sites are assessed based on their ability to contribute to the isolation of the waste and limit radionuclide releases to minimize potential adverse impacts on humans and the environment. Important site characteristics include[122]:

- Geology: The site is expected to possess a stable geology that contributes to the isolation of waste. In addition, the overall predictability of site evolution with regard to its performance in the future needs to be adequate.
- Hydrogeology: The hydrogeological characteristics of the site are expected to limit the contact between waste and groundwater, and thus minimize the mobilization and transport of radionuclides.
- Geochemistry: The geochemical characteristics of the site should be such that the potential for radionuclide migration is minimized. In addition, the chemical conditions should not adversely affect the durability and performance of the waste packages and engineered barriers.
- Seismicity: Seismic events expected to affect the site need to be assessed to ensure that the structures of the facility are designed and built in such a way that their performance will not be compromised.
- Topography: The geomorphological conditions of the site need to be such that surface processes, for example, erosion and flooding, are expected to be minimal in rate and intensity, or absent, precluding, therefore, the possibility that the ability of the site to isolate the waste might be compromised.
- Climate: The climatic conditions of the site need to be such that the isolation barriers can be expected to perform as designed for the required period of time. In addition, possible climatic variations need to be analyzed to ensure that the performance of the isolation system will not be adversely affected.
- Human environment: The implications of human activities in the area need to be considered.

Data collected during site investigation are used as input to the design process and to the associated safety and environmental assessments undertaken to determine site suitability. These data also provide baseline information on the undisturbed characteristics of the site, with which the characteristics of the site in the future can be compared, for example, in the context of confirming the adequacy of the models used to represent the behavior of the repository. Data collection is the most extensive activity to the site investigation process.

The siting process includes four stages, which are (i) conceptual and planning stage, (ii) area survey stage, (iii) site characterization stage, and (iv) site confirmation stage. Public involvement and participation represent one of the most important nontechnical aspects that affect the development of a disposal project. They vary in their nature and extent form country to country, depending on existing legal and political frameworks and on cultural context. Recent experience suggests that broad public acceptance will enhance the likelihood of the disposal approval. An important element in creating public acceptance is the perceived trust and credibility of the responsible organization and of the reviewing agency or agencies. Approaches have been developed to achieve improved credibility by giving the interested public a chance to express their views, by appropriate consideration of public comments, and by providing open access to accurate and understandable information about each phase of the disposal life cycle. The transparency and traceability of the decision-making process are important to generate public trust.[123]

3.4.1.1.2. *Disposal design*

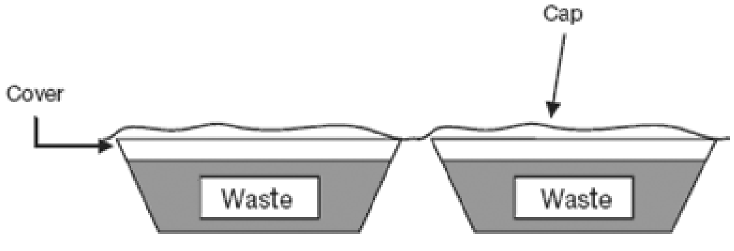
The disposal should be designed to provide adequate isolation of disposed waste for all times in the future. In this regard, adequate isolation means that any releases from the facility must be below health-based limits at any time in the future. Strategies to achieve this goal may be based on the performance of the engineered barriers or on the characteristics of the geological setting. The goal is for the overall disposal system to produce adequate performance, rather than an undue focus on any single part of the system. The design of disposal rely on the multi-barrier concept, in this concept the natural and engineered barriers systems components are used to isolate and retard radionuclide migration into the surrounding environment and to dilute the released radionuclides toward the biosphere.

The period of time for which performance credit can be taken for institutional controls is generally predefined by the regulatory authorities, taking into account the characteristics of the site, the relevant regulatory requirements, and various societal and ethical factors. The quantities and characteristics of the waste that needs disposal are also critical factors for the design of the repository. Based on the characteristics of the disposal site and waste and the anticipated duration of institutional controls, a detailed design of the disposal is developed. The detailed design may include a limitation of the hazardous content of the waste through definition of appropriate waste acceptance criteria. The disposal facility includes the waste emplacement area, buildings, and services for waste receipt. The design and layout of the site varies depending on the type, characteristics, and quantities of waste and on the site characteristics.

Disposal options are categorized as geological and near surface. Near-surface disposal includes two main types of disposal systems: (i) shallow facilities consisting of disposal units located either above (mounds, etc.) or below (trenches, vaults, pits, etc.) the original ground surface and (ii) facilities where the waste is emplaced at greater depths, usually in rock cavities. Geological disposal refers to disposal at greater depths, typically several hundreds of meters below ground. Figure 16 presents various design concepts for radioactive waste disposal.[124–126]

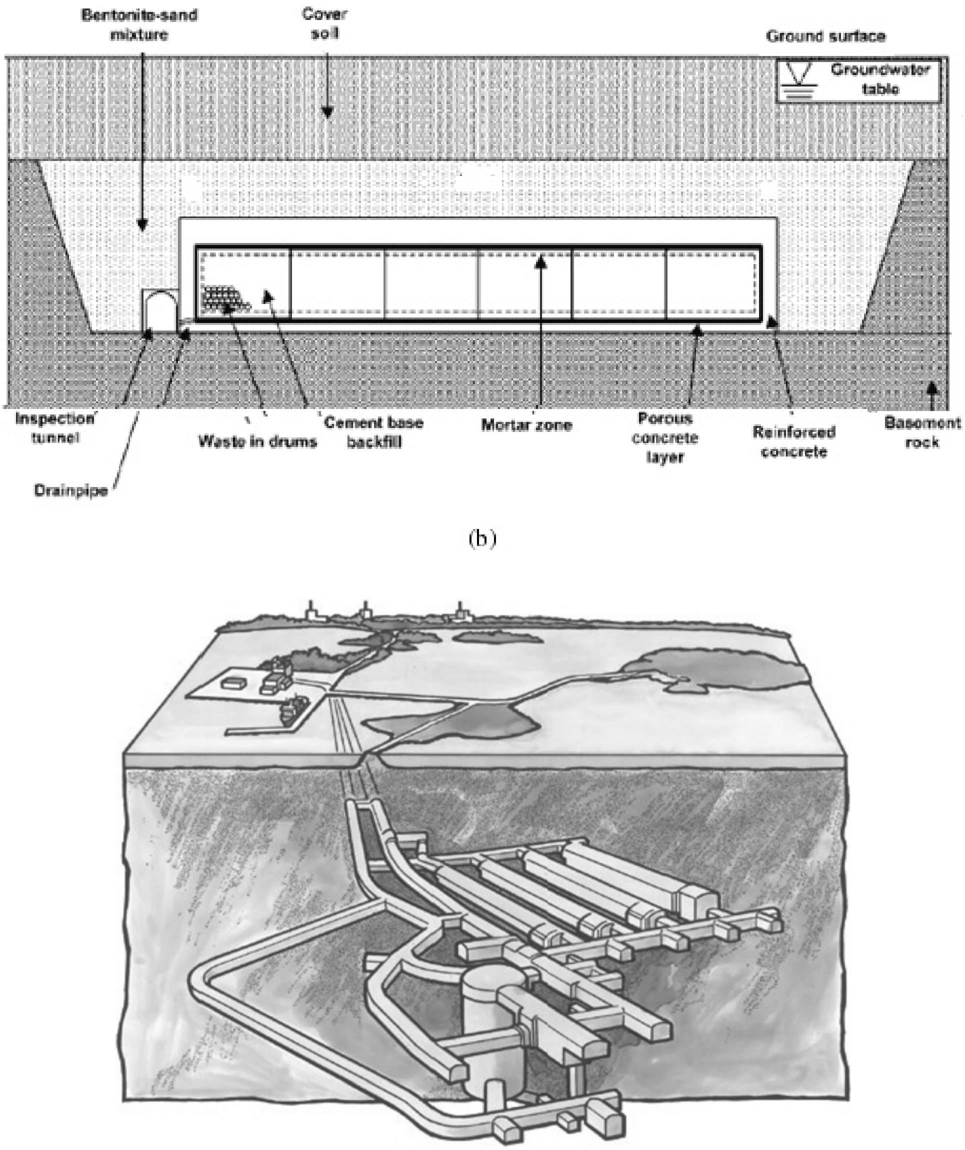
Different shallow disposal designs have been practiced in various countries; these designs could be classified as:

- The covered trench concept is the oldest and simplest of the disposal concepts, which consists of placing waste into excavated trenches and covering the filled trenches with soil. Disposal sites using this concept frequently have retrofitted engineered barriers. This concept has been applied at Drigg disposal facility in the United Kingdom.
- The closed vault concept consists of a concrete vault into which is placed packaged and/or treated waste. The void may be backfilled and the structure closed with concrete slabs, which may be sealed by, for example, asphalt. The whole structure is then protected by an earthen cap. Examples of the application of this concept in Centre de l'Aube in France, El Cabril in Spain, and Rokkasho-mura in Japan.
- The free-draining vault concept in which infiltration is controlled by placing waste in a dry permeable layer and covering the waste with an impermeable concrete



(a)

Figure 16. Disposal Options: (a) Simple Disposal Trench,[124] (b) Shallow Land Disposal Vault,[126] and (c) Geological Disposal,[125] (d) Borehole Disposal.



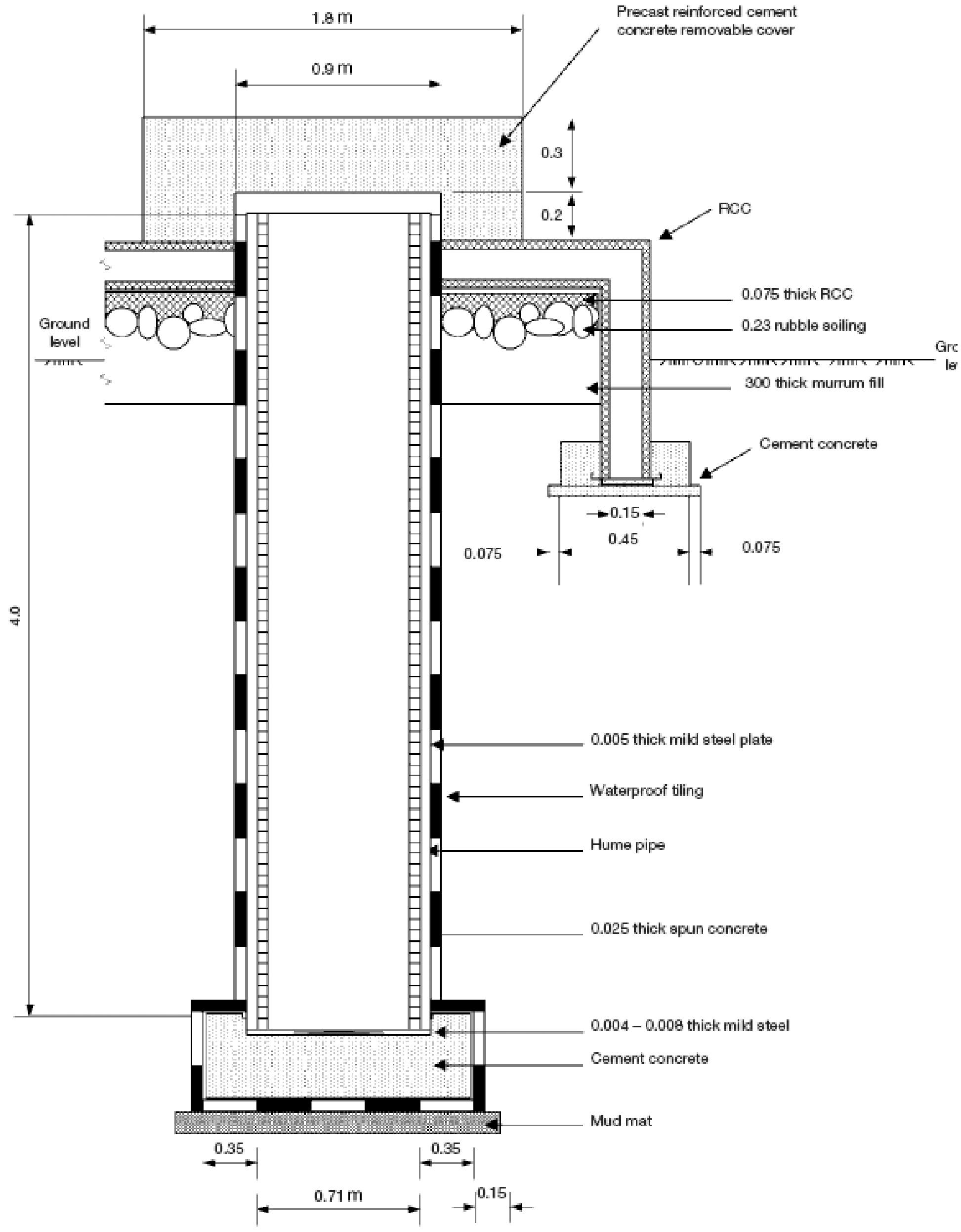
(b)

(c)

Figure 16. (*Continued*)

roof that is subsequently protected by an earthen cap. This disposal concept has been applied in IRUS disposal facility in Canada.

- The open vault concept has a low-permeability cap that placed over the filled vault without emplacement of a concrete slab. Waste is, however, pretreated to minimize void. The cap is designed to accommodate some settlement.



(d)

Figure 16. (*Continued*)

Facilities built to greater depths include specially excavated caverns tens of meters deep below the Earth's surface, disused mines, natural cavities, and borehole. The borehole disposal concept includes the emplacement of conditioned radioactive waste in a narrow diameter engineered facility operated directly from the surface. Borehole disposal facilities cover a range of design concepts with varying depths and diameters.[127]

Deep disposal, several hundred meters below the surface in stable geological formations, is generally recognized to be an appropriate solution for radioactive waste arising from nuclear power generation, high-level waste from reprocessing operations, spent fuel elements (when considered as waste), and alpha bearing waste.

3.4.1.1.3. *Waste acceptance criteria*

The waste acceptance criteria are predetermined specifications that establish requirements for the waste form and waste packages for disposal in a specific facility,[128] which are established directly by the regulatory authority or by the implementing organization approved by regulatory. These criteria should be derived from consideration of both operational requirements and accidental situation and they should be quantitatively or qualitatively based such that conformance can be either assessed by direct measurement and/or assured by the application of appropriate management methods and controls during the conditioning process.

Compliance with waste acceptance criteria generally requires the definition of a waste package's characteristics and attributes, including performance data (compressive strength, load bearing capability, resistance to impact, corrosion, fire resistance, etc.) and identification of quality-related parameters that need to control to provide assurance of conformance with waste acceptance criteria. In addition, confirmation of the conformance of individual waste packages to the requirements of a waste package specification. Table 16 lists the key waste acceptance criteria that need to be addressed by both the waste package specification and the waste package data sheet.[129]

3.4.1.2. Operational phase

During this phase, the disposal is open and operational. Waste packages, complying with the waste acceptance criteria established during the preoperational phases, are received and placed into disposal units; any auxiliary conditioning/packaging facilities and all supporting units are operating.

At the end of the operational period, appropriate steps are taken to permanently close the disposal. Facility buildings are decommissioned and, if contaminated, may be disposed of in the facility. Closure systems are usually constructed. Appropriate institutional controls are put in place prior to disposal closure. These controls can

Table 16. Identification of Criteria to Be Addressed in Waste Acceptance Criteria.

Waste Acceptance Criteria Item	Waste Package Specification (General Demonstration of Compliance)	Waste Package Data Sheet (Individual Demonstrations of Compliance)
Quality assurance program	A description of the arrangements established by the operator to ensure the effective management and control of all parameters identified as critical to achieve the waste package quality. This description should cover all activities, events, and resources necessary to ensure compliance with these specification.	Certification by those responsible for the operation of conditioning facilities or the conformance of individual waste packages with requirements identified in the waste package specification.
Compliance with statutory and regulatory requirements	Arrangements to ensure compliance with identified statutory and regulatory requirements should be included within the quality assurance program.	
General description of the raw waste — Physical, chemical, and biological properties — Radionuclide content — Fissile content	• A written description of the waste should be provided with the details of its source, volume, and weight. • The characteristics and composition of the raw waste should be identified and where possible quantified with sufficient accuracy. • Limits should be established for those radionuclides and other properties that could adversely affect the suitability of a waste package for disposal where appropriate. • For all components present in the waste, average and limiting values for an individual waste package should be defined along with a maximum limit of fissile content to safeguard against criticalities.	• Confirmation that the raw waste content of a waste package is within the limits defined in the waste package specification should be given. • Where fissile material is present, then it should be confirmed that it is within the permitted fissile mass limit.

(*Continued*)

Table 16. (*Continued*)

Waste Acceptance Criteria Item	Waste Package Specification (General Demonstration of Compliance)	Waste Package Data Sheet (Individual Demonstrations of Compliance)
Description properties weight	• A description of the waste container to be used to hold the conditioned waste should be provided, using drawings where possible, along with details of its mechanical and physical properties. • Reference to a manufacturing specification should be included and the weight of the container empty given.	• Confirmation that the container conforms to the requirements of the manufacturing specification should be given, with reference to a manufacturing release certificate where possible (to provide access to the manufacturing records if necessary) and details of any manufacturing concessions granted.
Details of any immobilizing matrix used including the matrix specification and arrangements for control of quality-related parameters	A description of any matrix used to immobilize the raw waste or elements within it, comprising: — identification of the waste to be immobilized, — details of any pretreatment earned out on the waste, — specification for the immobilizing matrix including its composition ratio rate to immobilized product percentage voidage and degree of homogeneity within the final product compressive strength, — results of tests to assess the acceptability of final product for leaching and release of included radionuclides, and — identification of those parameters where need to be controlled in order that the final product conforms to the specification and a description of the arrangement for monitoring and controlling them.	Confirmation that any immobilizing matrix used meet the requirements of the specification concerned should be given.

(*Continued*)

Table 16. (*Continued*)

Waste Acceptance Criteria Item	Waste Package Specification (General Demonstration of Compliance)	Waste Package Data Sheet (Individual Demonstrations of Compliance)
Package type and variant demonstration of package integrity covering: — Mechanical strength, — Resistance to impact, — Radiation stability, — Fire resistance, — Voidage, — Durability, — Resistance to teaching package identification, labeling, and marking.	• The package type and variant should be defined. • The results of work carried out to assess and demonstrate the integrity of the waste package against each of the identified requirements should be reported. • The system to be used to mark/label packages should be described along with the details of the assignment of a unique identifier (numeric, alphanumeric, or bar code) to each package.	The package type and variant should be recorded on the data sheet. The package identifier should be clearly marked on at least two faces of the container and recorded on the data sheet.
Package weight	• The maximum weight of an individual package should be specified and shown to be compatible with any limit specified in the waste acceptance criteria and/or requirements for the handling of waste packages. • The method of determining this limit should be defined. • Arrangements for determining the weight of each package or verifying that it is within the limit specified should be described.	The weight, surface contamination levels, and dose rate of each package should either be measured or confirmed as being within the limits given in the waste package specification, and recorded on the date sheet on completion of conditioning.

(*Continued*)

Table 16. (*Continued*)

Waste Acceptance Criteria Item	Waste Package Specification (General Demonstration of Compliance)	Waste Package Data Sheet (Individual Demonstrations of Compliance)
Dose rate	• The average and maximum dose rates for an individual package should be specified. • The method of determination should be defined and the specified dose rates shown to be compatible with any limit specified in the waste acceptance criteria and/or handling, storage, and transportation requirements. • Arrangements for determining the dose rate of each package or verifying that they are within the limits specified should be described.	
Storage	Where a package is to be held in interim storage prior to disposal, then a description of the storage facility and the arrangements for monitoring the condition of packages and minimizing any deterioration in their condition should be described.	
Transport	• Arrangements for preparing the package for transport to the repository and demonstrating compliance with national requirements should be described. • Any documentation in addition to the package date sheet required by those regulations should be identified.	Compliance with national (and where necessary international) regulations governing the transport of radioactive materials should be confirmed on the relevant documentation.

(*Continued*)

Table 16. (*Continued*)

Waste Acceptance Criteria Item	Waste Package Specification (General Demonstration of Compliance)	Waste Package Data Sheet (Individual Demonstrations of Compliance)
Consignment documentation	• All documents and records to be handed over to the repository operator when a package is consigned for disposal should be identified and samples attached, where necessary. • Except where the consignment documentation used is that specified in the waste acceptance criteria, it should be shown how the documentation concerned fulfills and meets the requirements of those criteria. • A description of the arrangement for completing this document should be given, along with the details of those personal authorized to sign them.	Completion of consignment documentation as described in the waste package specification.

enhance both the long-term safety of the disposal and public confidence in its long-term performance. Institutional controls may include active controls, such as monitoring, surveillance, and remedial work, and passive controls, such as land use restrictions and record keeping.

3.4.1.3. Post-closure phase

During this phase, following closure of the disposal facility, the site is maintained under institutional control. Access to the site is controlled and a monitoring program, approved by the regulatory body, is implemented. Controlling access to the site is important in that it serves to minimize the potential for human intrusion. During this period, any perturbation of the disposal system revealed by the monitoring and surveillance program can be investigated and appropriate remedial actions can be taken. At the end of the institutional control period, it is expected that the radioactivity in the waste will have decayed to sufficiently low levels that the radiological risk to an inadvertent intruder into the waste is acceptably low. The duration of both the active and passive institutional controls needed to ensure safety depends on many factors, such as the waste characteristics, site characteristics and facility design, and economics. However, institutional controls for near surface disposal facilities are generally considered to be effective up to at most a few hundred years.

3.4.2. *Safety Assessment*

The long-term safety of radioactive disposal must be convincingly demonstrated before its implementation. Assessment of the safety impact arising from the disposal is a main tool to investigate and explain the long-term behavior of a disposal facility. It is the only existing approach for linking observable features of the site with the design goal: the safety of the disposal system in the future.

A summary of the hierarchy of terms commonly used relating to the assessment of the safety impacts arising from the disposal of radioactive waste is given in Table 17. At the top of the hierarchy is the safety case, which uses the results from the safety and performance assessments of the repository linked with other factors that are important for the assurance of safety such as the use of sound science and engineering, QA procedures, safety culture, robustness and defense in depth, and institutional controls.[130,131] At the next level, there are the performance and safety assessments. Performance assessment involves an analysis of the performance of a system or subsystem, followed by comparison of the results with appropriate performance criteria. In contrast, safety assessment is the analysis of the overall system and its impact, followed by comparison with appropriate safety criteria. The performance and safety assessments are, in turn, underpinned by performance and safety analyses. The results of a safety assessment can be presented in a way

Table 17. Hierarchy of Commonly Used Terms Relating to the Assessment of Radioactive Waste Disposal.[130]

Safety case	
It includes performance and safety assessments. In addition, a full line of arguments and evidence that a sufficient set of processes have been analyzed and appropriate models and data used; relevant overall measures of performance and safety are within acceptable ranges, allowing for uncertainties. More qualitative and parallel lines of evidence and reasoning may be also used to support results of the quantitative modeling and to indicate the overall safety of the system.	
Performance assessment	*Safety assessment*
It includes performance analysis. In addition, comparison of intermediate parameters with appropriate criteria set by regulation design targets.	It includes safety analysis. In addition, testing of arguments that a sufficient subset or of processes have been analyzed, appropriate models and data used, plus comparison of calculated measures of overall performance to regulatory safety limits and targets.
Performance analysis	*Safety analysis*
Quantitative analysis of some subset of processes relevant to the behavior of the disposal system and calculation of intermediate parameters of interest.	Quantitative analysis of the set of processes that have been identified as most relevant to the overall performance of the disposal system and calculation of a measure of overall performance relevant within the given national regulatory regime.

that provides reasonable assurance of the performance of individual system components. Thus, performance assessment can be viewed as an integral part of safety assessment.

There are various assessment methodologies that have been developed to assist in developing an appropriate safety assessment. While there are differences in the detail of the approaches used, safety assessment methodologies have the following key components.[132,133]

- assessment context specification,
- disposal system description,
- scenarios development and justification,
- model formulation and implementation, and
- result analysis of and building of confidence.

Safety assessment of a radioactive waste disposal facility is generally undertaken to provide confidence to government, regulatory authorities, general public, and technical/scientific personnel that the facility will ensure the safety of people and protection of the environment over long timescales. However, this generic objective does not provide a very precise description of what has to be considered in the assessment. The assessment context provides a framework for the performance of the safety assessment, establishing the purpose, the regulatory framework, the assessment end-points, the assessment philosophy, the disposal system characteristics, and the timeframes of concern. The assessment context is intended to provide clear and comprehensive goals for the analysis.

The disposal system is frequently considered to consist of the near field, the geosphere, and the biosphere. The near field includes the waste, and the disposal area, and the engineered barriers of the disposal facility, including the disturbed zone of the natural barriers that surround the disposal facility. The geosphere includes the rock and unconsolidated material that lies between the near field and the biosphere. It can consist of both the unsaturated zone (above the groundwater table) and the saturated zone (below the groundwater table). The biosphere is the physical media at the point of discharge from the geosphere (atmosphere, soil, sediments, and surface waters) and the living organisms (including humans) that interact with them. For the purposes of safety assessment, it is necessary to have a sufficiently clear description of the disposal system that facilitates the development of the safety assessment, ensuring that key features, events, and processes that may affect disposal system safety are adequately described. Since the biosphere is particularly susceptible to the actions of future humanbeings, and since those actions are impossible to predict, assumptions about the biosphere are usually stylized.[5,134]

In a safety assessment of a waste disposal facility, it is important to assess the performance of the disposal system under both present and future conditions, including anticipated and less probable events. Different countries have various requirements for the extent to which low probability events need to be included in the safety assessment. Taking such events and processes into account means that many different factors should be included in the safety assessment and evaluated in a consistent way, often in the absence of quantitative data. This is often achieved through the formulation and analysis of a set of scenarios, which are stylized representations of the potential future conditions that may affect the disposal facility. The scenarios generally need to represent a reasonable set of conditions of concern that may influence the future safety of the facility. However, practical considerations dictate that the number of such scenarios must be limited. The tradeoff between the desire to be comprehensive and the need for limiting the scope of the safety assessment is a primary focus of regulatory reviews of safety assessments.

Once the scenarios have been developed, their consequences in terms of the assessment context are analyzed. Depending on the nature of the scenario, an appropriate approach for its analysis is chosen. For some scenarios, it may be appropriate to use a qualitative assessment approach (e.g. when data are not available). For the scenarios that are to be quantitatively assessed, the scenarios should be organized into a form that can be mathematically represented. A set of model-level assumptions (about dimensionality, boundary conditions, etc.) are needed for each of these scenarios. These assumptions comprise the conceptual model. More than one conceptual model may be consistent with available information for a scenario.

Once the scenarios and associated conceptual and mathematical models have been developed and implemented in software tools, and the associated data collated and implemented, calculations can be undertaken to assess the impacts of disposal facility. The results then need to be collated, analyzed, and presented. The nature of disposal facilities and the requirements for projecting doses long times in the future inevitably leads to uncertainties. To be credible, safety assessments need to provide a sufficient representation of the uncertainties that a skeptical outsider (usually a regulator) can be convinced that the site will be safe. The standard of proof sought in waste disposal safety assessments has been variously called "reasonable assurance" or "reasonable expectation." These terms are invoked to emphasize that absolute assurance is not normally achievable for analyses that project consequences over very long times. Issues of the uncertainties in safety assessments and their special consideration have been discussed in the literature.[18]

Establishing confidence in the disposal development program is an important consideration at all stages of the disposal life cycle. It is particularly important that government, regulatory bodies, public, and technical/scientific personnel should have confidence in the program. A variety of measures can be used to help establishing confidence.[135–138] These include:

- Application of a systematic approach to the disposal program and its associated steps; this approach should have a number of confidence enhancing measures such as measure to insure that each stage of the program and its associated decisions are appropriately and clearly documented; utilization of transparent scientifically and technically justifiable methods, and apply multiple lines of reasoning.
- Peer review of the program as well as of its individual stages; it is an important activity that can be used to achieve scientific and technical confidence in the approaches, methods, data, and decisions made in the development of a disposal facility.
- Demonstration of the robustness of particular aspects of the program; safety assessment have a key role in the demonstration of the site and of the design robustness through the provision of a basis for rational and technically sound decisions relating to their safety. Further confidence can be developed in the design through the use of multiple barriers concept. Additionally, field tests and monitoring are undertaken to demonstrate the appropriate performance of the barriers. Operational monitoring data can be used to confirm that the facility performs according to the design objectives.
- Identification and management of uncertainties; uncertainties arise from the disposal system evolution over the timescales of interest, uncertainty in the

conceptual, mathematical, and computer models used to simulate the behavior of the system and uncertainty in the data and parameters used as inputs in the modeling. In order to build confidence to assist the decision-making process, it is important that these uncertainties are identified and managed appropriately.

- Application of QA procedures throughout the program; an important contribution to building confidence in a repository safety case arises from the application of a QA program.
- Documentation of the disposal program and the preservation and availability of the associated documents are very important to generate confidence in the safety of disposal.
- Involvement of the public in decisions relating to the program.

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Chapter 17

PLASTICS WASTE MANAGEMENT IN INDIA: AN INTEGRATED SOLID WASTE MANAGEMENT APPROACH

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Abstract

India has witnessed a substantial growth in the production of plastics and an increased consumption of plastic. In the absence of adequate waste collection and segregation process, the management of the waste created by discarded used plastics items, especially ones used for packaging applications has become a challenging task. This article provides an overview of the resource recovery from plastic waste with consideration of integrated waste management (IWM), to evaluate the best possible option for tackling waste in Indian circumstances.

Keywords: Plastics, polymer, waste management, recycle, energy recovery, incineration, IWM, fuel.

1. Introduction

Economic development significantly contributes to improvements in life standards. Therefore, both economic development and environmental conservation are the immense important aspects and priorities of 21st century. Both require simultaneous indispensable support and adequate consideration, so that they are in fact not only being compatible but also remain mutually supportive. However, coupled with life standard improvement, economic prosperity also induces environmental degradation with long-term irreversible consequences for nature. Rapid population growth, urbanization, and industrial growth have led to severe waste management problems in several cities around the world. Simultaneous development in economic prosperity and industrialization often conflict with sound environmental considerations. The real problem, however, is the lack or inadequate environment management at a grass root level. The basic requirement is, therefore, need an approach toward technological development for the minimization of environmental degradation.

Plastic as a synthetic polymer substitute natural materials in almost every aspect of our life and become an essential part of our society. Nature has witnessed a considerable intensification in the production of plastics in last few decades and simultaneous increased consumption of plastic materials. With time, stability and durability of plastics have been improved continuously, and hence these groups of materials are now considered as a synonym for materials being resistant to many environmental constraints.[1] The basic properties *viz.* durability, resistance to chemicals, safety and hygienic nature, relative inexpensiveness to produce, thermal and electrical insulations, and lighter weight than the competing materials helped plastics to be indispensible in every aspects of life. Plastics compromise diverse group of chemically complex compounds. Plastics are formed into any number of products, and different plastic resins are difficult to differentiate. This leads to problems in collection, separations, and recycling. Because of its durability, plastics accumulate and remain persistent in the environment at the rate of 25 MT per year.[2] Moreover, converting plastics down to their original chemical constituents is often technologically infeasible or otherwise unprofitable. Management of plastics found in municipal solid waste (MSW) is most critical sector because of continuous increase in plastic proportion in MSW, its nonbiodegradability, and direct harmful effect to society.[3]

Basically, problems related to solid waste persist beyond merely its disposal. In addition to technical and environmental complications, there are administrative, economic, and societal tribulations that must be addressed. The scientific efforts to sort out all these complications are usually referred as waste management. In this aspect, the management encompasses planning, design, and operation of facilities for collecting, transporting, processing, recovering, and finally disposing of waste.[3] Waste management and disposal is the most neglected sector in India, and about 90%

of waste is currently disposed of by open dumping.[4] There are lacks of efficient waste collection, segregation, and treatment facilities in most parts of India, even in the most developed cities. Some commonly used methods by which the waste could be managed are incineration, landfilling, and composting. However, all these methods are practiced in unscientific manner and precautions regarding safe disposal of waste residues are not taken care off. The basic properties that have helped plastics to be most useful in common life are also cause of concern in environmental safeguard perspectives. Waste plastic thrown on land mostly enter into municipal drainage lines and chokes it resulting into floods as experienced in Mumbai, India in 1998. Again, millions of mammals, birds, reptiles, and fish are reported to be killed every year by the ingestion of plastic bags. Mostly plastics affect marine wildlife either by entangling creatures or by being eaten. Turtles are particularly badly affected by plastic pollution, and all seven of the world's turtle species are already either endangered or threatened for a number of reasons.

Productive use of waste represents a means of mitigating some of the associated problems of solid waste management. The concept of integrated solid waste management (ISWM) is meant to provide a sustainable framework both for manufacturers and consumers.[3] The ISWM is intended to guide decisions about the generation of wastes, recycling of materials and ultimate disposable of residues.[5] It helps to save and sustain natural resources that are not replenished, decreases environment contamination, and serves to save and recycle energy production processes. Wastes should be considered as potentially valuable resources merely awaiting appropriate treatment and application.[6] Unscientific disposal of plastic wastes triggers environmental degradation due to their long biodegradation period, therefore, logical methods for reduction of their negative effects should be application of ISWM concept for its effective management with optimum recourse recovery.

2. Global Scenario of Waste Plastics

2.1. *Plastic Production, Consumption, and Waste Generation: Global Scenario*

Globally, each year nearly 140 MT of plastics is produced.[7] A recent study in Western Europe estimated the annual total consumption of plastics at 49 MT (in 2003) corresponding to 98 kg per capita.[8] Decadal growth (1993–2003) of per capita annual plastic consumption in Western Europe was 34 kg. The global plastics additives market was about 9.9 MT in year 2000 with a value of US$19 billion. Nearly, 80% of the global plastics additives were being consumed by the USA, China, India, and Eastern Europe outside of the European Union. However, South East Asia, especially India and China, has emerged as the global leader in plastics consumption, with

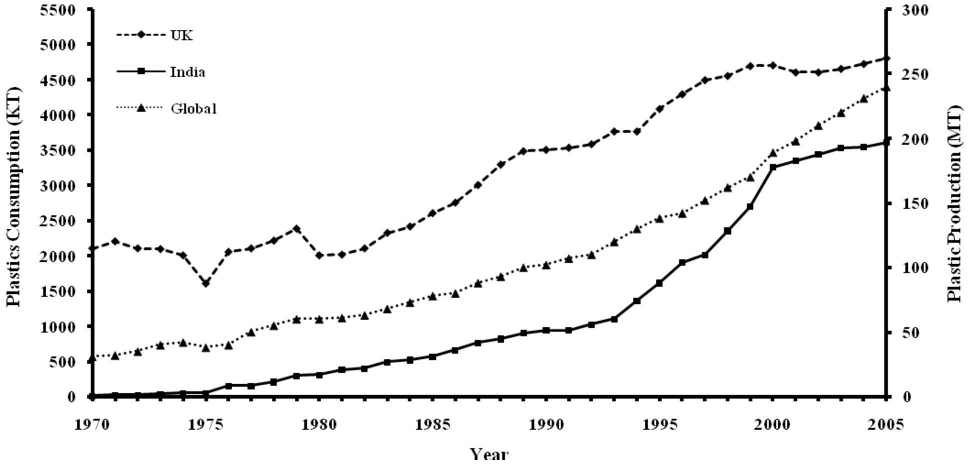

Figure 1. Comparative Study of Global Plastic Production (MT) with Consumption (KT) in India and UK.

over 52 MT consumption of plastics in 2004. Plastic additive markets are growing at about 3% annual rate in Europe and Asia, whereas China is predicted to grow at 8–10% (www.plastemart.com).[9] The annual consumption of plastics in the USA is estimated as 38.9 MT, closely followed by China 38.8 MT per annum. India is also projected to be the third largest consumer market for plastics in 2009 with a total annual consumption of 12.5 MT. In the 1990s, plastic consumptions in India grew exponentially with an average growth rate of 12%.[10] Current growth rate in India's plastic consumption is also predicted higher than that of China and any other developing countries and well comparable to that of UK (Fig. 1).

2.2. *Plastic Production, Consumption, and Waste Generation: Indian Scenario*

In 1990–1991, India produced 0.363 MT of plastics polymer, but within a decade, an incredible 890% increase leads to total plastics production to 3.2 MT (2000–2001). Plastics production in India further rises to 4.77 MT in 2005–2006, maximum of which are polypropylene (PP) and high-density polyethylene (HDPE). Among different types of plastic polymer, low-density polyethylene (LDPE) demonstrates maximum growth in consumption in India closely followed by HDPE and PP (Fig. 2). Polyethylene (PE), PP, and polyvinyl chloride (PVC) also contribute a large share in India's polymer market mainly due their low cost and durability. On an average, the commodity plastics *viz.* PE, PP, PVC, and polystyrene (PS) accounts 80% of the total plastic consumption in India (Fig. 3). In 1990–1991, per capita consumption of plastics in India was 0.8 kg but within a decade, per capita consumption significantly

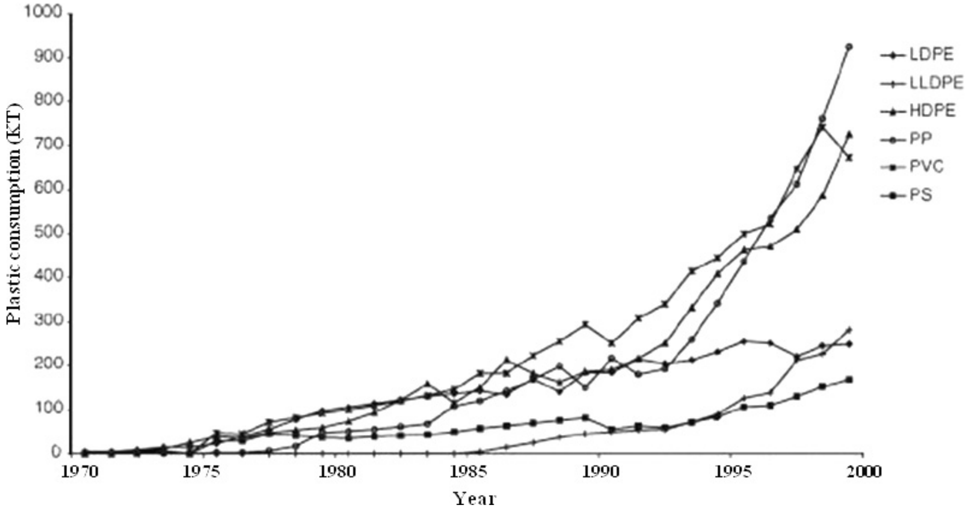

Figure 2. Average Growth Rate of Virgin Plastics Consumption in India.[3,10,12,40]

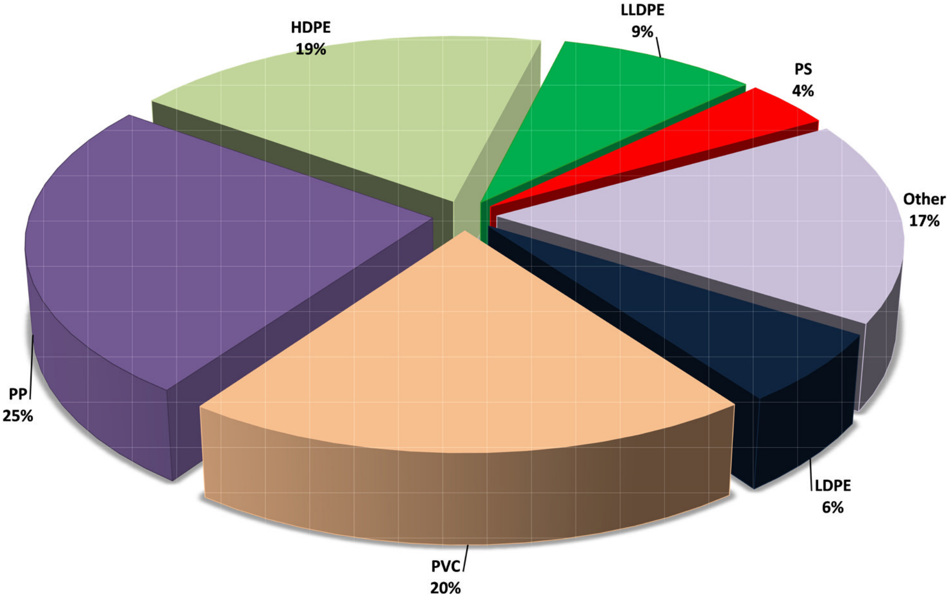

Figure 3. Consumption of Different Virgin Plastic Resins in India.[40]

increases to 3.5 kg (2000).[3] However, it is still far below than the global average (18 kg).[10] However, the projected estimates of per capita plastics consumption in 2021 may reach to a substantial figure of 10.9 kg,[11] which seems a realistic considering the rapidity with which plastics are replacing its competitive materials.

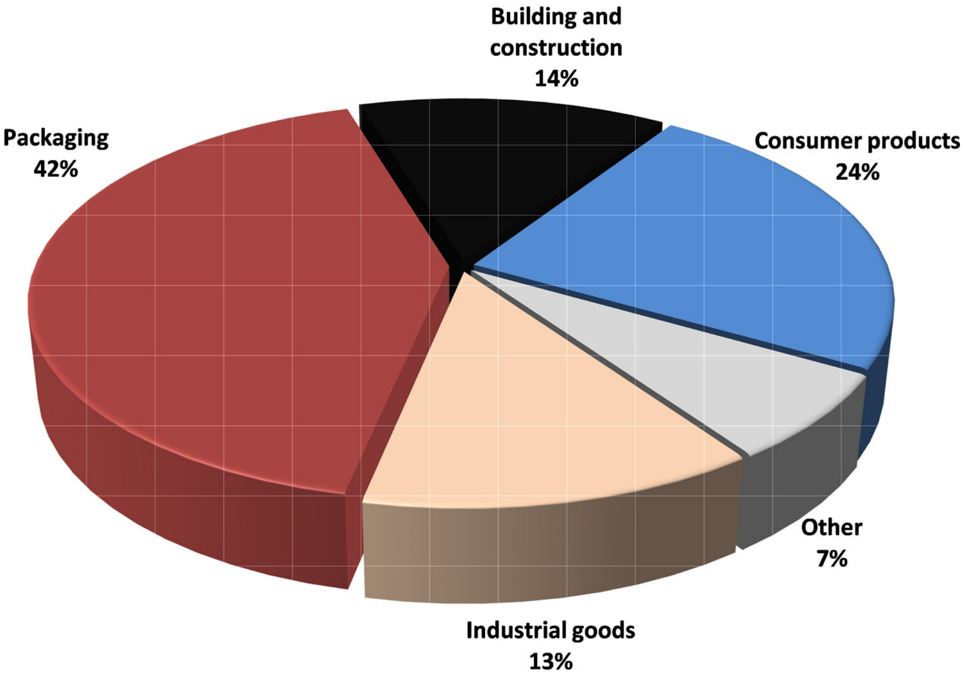

Figure 4. Percentage of Plastic Consumption in India by Different Market Sector.[12]

Packaging represents the largest single sector of plastics use in the India. The sector accounts for 42% of plastics consumption and plastic is the material of choice in nearly half of all packaged goods.[12] Apart from use in packaging, plastics are also extensively used in the consumer products such as furniture and housewares, building and construction, and in industrial sectors (Fig. 4). However, according to research findings of National Plastic Waste Management Task Force,[13] packaging constitutes 52% of the total India's plastic consumption. This is line with consumption pattern of other countries such as the USA and UK, where packaging exhibit maximum share in total plastic consumption. After primary use of this portion of plastics, annually 0.93 MT of waste plastics are discarded along with the household waste.[11] Although, rag pickers recover a considerable portion of this waste, but considerable amount of it either being soiled with the organic matter or not found appropriate for further processing. In India, PE, PP, and PVC dominate the market mainly because of its low cost, chemical structure, physical advantages, and high durability. Polyolefins account for 60% of the total plastic consumption in India.[10]

Annually 1.3 MT of plastic waste is generated in India, which is 36% of total India's plastics consumption. Nearly, 42% of total generated plastic waste is recycled in India by 20,000 recycling industries with total potential of 0.37 MT/annum. According to NPWMTF (1997), in 2000–2001, more than 5,400 tonnes of plastics waste being generated in India per day. Percentage of plastics in MSW has also increased significantly from 0.7% in 1971 to 4% in 1995.[14]

Lack of biodegradability of commercial polymers, particularly used in packaging, industry, and agriculture, focused public attention on a potentially huge environmental accumulation and pollution problem that could persist for centuries. Disposal of plastic wastes has potential harmful effects on the environment and, therefore, the logical methods should be to recover maximum energy to maintain environmental sustainability. ISWM is the concept to formulate decisions about the generation of wastes, recycling of materials, and ultimate disposal of waste residues.

3. Integrated Solid Waste Management

Solid waste treatment and disposal methods are burdened with problems. The collection of solid waste and their transport to treatment facilities or to landfills accounts for roughly three-fourth of the total cost of waste management. Health and hygienic issues are also associated with transportation and, therefore, require special attention. Landfill sites are mostly prone to soil and groundwater contamination, unless scientifically managed. Recycling of waste materials also preferred in some aspects but do possess technological constraints coupled with chances of future contaminations. Incineration of waste materials has had problems with odor and air pollution and also may not be found feasible due to intrinsic properties of the waste material. The priority of waste management policy is to reduce the quantity and toxicity of waste. The concept of waste minimization has been also widely implemented in recent years. The role of waste prevention can be suitably illustrated in Fig. 5. Together with waste prevention, significant waste reduction can be achieved by inducing the concept of changing product that should focus on pollution reduction and resource efficiency and implementation of green design concept. Green design mainly concerns with the reducing the environmental impacts associated with the collection and processing of raw materials, manufacturing, product use, and disposal of the product (Fig. 6). It is an important part of the waste and pollution prevention strategy. According to the Office of Technology Assessment,[15] major components of the green design are waste prevention and better management of materials. Moreover, once a particular product reached to its end of life, the materials still possess some secondary economic values and, therefore, additional savings can be made by reducing easy disposal. Green design eases the process by which secondary raw material can be retrieve from any product. Moreover, a critical stage in developing a product is the selection of appropriate materials and, therefore, attempts should be made to select such raw materials and technology that are economically feasible and also environmentally sustainable. An important aspect of raw materials selection is the need to

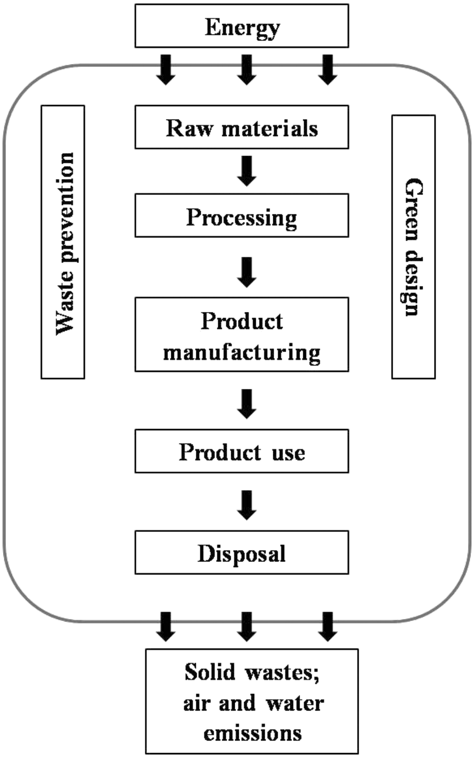

Figure 5. Schematic Representation of Waste Minimization and Waste Prevention.

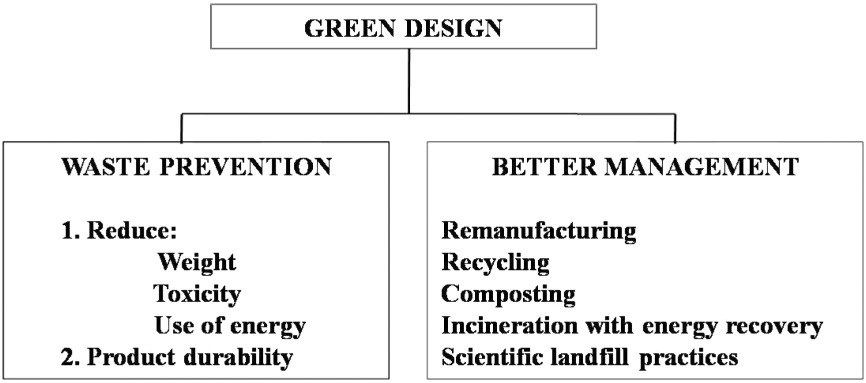

Figure 6. Schematic Representation of Green Design.[15]

reduce the toxicity of materials wherever possible. These toxic ingredients mostly create enormous problems during the waste management stages when the product loses its usability. Therefore, green design strategies help to reduce the amount of toxicity associated with any compound without hampering its usefulness and quality.

3.1. *Integrated Waste Management (IWM) and Resource Recovery: Basic Concepts*

Definition of waste has a significant effect on waste management. To define a material as a “waste” has an impact on what measures can be taken, and also on what measures are not permitted, as well as the administrative procedures applying to its transport, export or processing, sale, and reuse.[16] In general, waste is the material perceived to have little or no value. However, waste can be properly defined as the material that devoid of its primary economic value but possess secondary intrinsic value. Waste minimization and its effective treatments is the most challenging field in sustainable environment. Varieties of wastes are produced from different sectors and due to continuous technological advancement in the processing sector, it is expected that the characteristics of wastes are also changeable. It is mostly emphasized that many of conventional waste problems can be solved by minimizing the quantities of these materials and also through product substitution, waste recovery, recycling, and waste minimization.[3] The waste hierarchy as illustrated in Fig. 7 describes some conventional approaches for the both minimization and management of waste. In the hierarchy of the waste management, waste prevention is the most preferred option. Waste prevention is also described as the combinations of source reduction coupled with reuse of materials. It inherently minimizes the portion of a product to be discarded after primary use and also motivate to reuse of a particular product in different applications. Source reduction can be effectively implemented by designing and manufacturing products and packaging with minimum volume and toxic content so as to help ensure that the product has a longer useful life. Reuse follows source reduction in waste management hierarchy. It is a waste reduction strategy where a

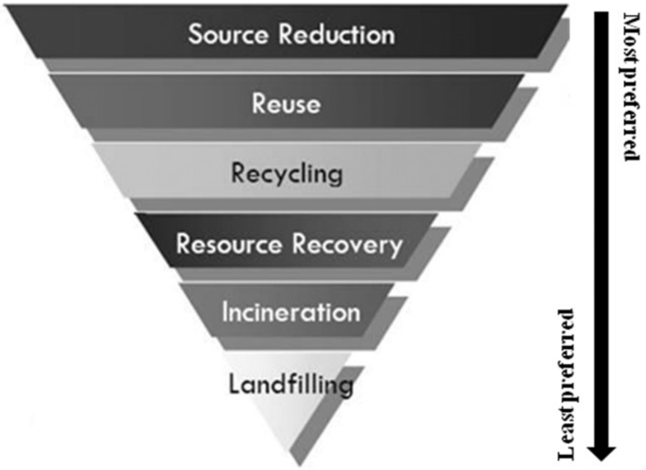

Figure 7. Hierarchy of Waste Management.

product is used for the same or new purpose without undergoing a physical change. Reusing, when possible, is preferable to recycling because the item does not need to be reprocessed before it can be used again.

After waste prevention, the recovery of materials for recycling and composting is given the highest priority. Recycling is somewhat different from reuse or remanufacturing and it defines the use of collected waste materials to be used as a raw material for a new compound. It provides the opportunity to reclaim valuable resources and to minimize the amount of waste placed in landfills. Depending upon the characteristics of the plastics, it may be recycled or not, but plastic recycling process do contain some technological constraints due to probable contamination and, therefore, special attention is required. The practice of the three R's (reduction, reuse, and recycle) fits very well within the sustainable development concept. Resource recovery, incineration, and landfilling are the less preferred options of waste management hierarchy, as they contradict the waste prevention concepts. However, there are potentials to recover huge amount energy and resources from the MSW if proper segregation and technology are available. Despite of having some advantages, the hierarchy of waste management has some limitations:

1. It is of little significance when combinations of waste management options are used for waste treatment.
2. It also does not consider the economic aspects of the waste management options and, therefore, it has some restricted applicability in different scenario.
3. Hierarchy does not address the specific local situations. As example, incineration of MSW in India is not economically feasible, as MSW mostly comprises the biodegradable compounds, which possess lower calorific value.

Therefore, rather than relying on a waste reduction hierarchy, IWM suggests the optimization of the system and also consider that all options can have a simultaneous role to play. The IWM uses a combination of techniques and approaches to handle targeted portions of the waste stream. It is important to realize that the portions of the hierarchy interact with each other and that change on one level will impact or influence another level. Moreover, using a range of waste management options in an integrated system gives the flexibility to choose the best possible waste management option suitable in particular situations.

3.2. *MSW and its Management*

Waste is an inevitable product of the society. In general, solid wastes other than hazardous and radioactive materials are considered as MSW, mostly comprises of all solid and semisolid materials discarded by a community. *Refuse* is the fraction of the MSW produced in a domestic household. The components of refuse are garbage or food waste, rubbish, glasses, cans, and papers including trash. Although all the

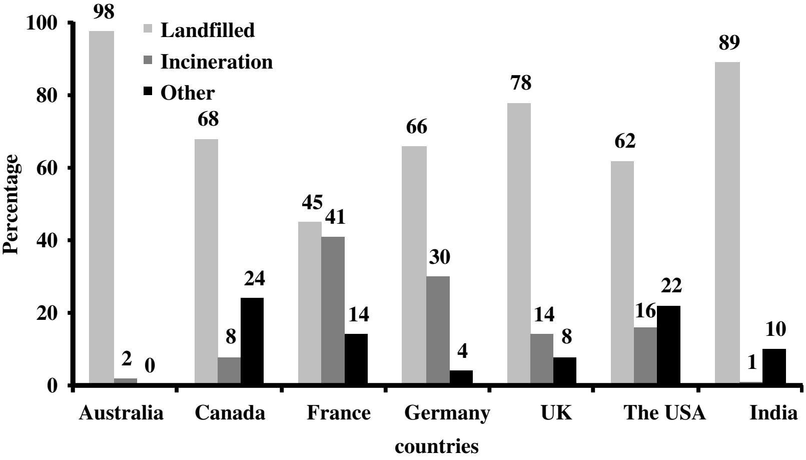

Figure 8. Comparison of MSW Disposal Processes in Different Countries.[42]

solid waste from residential, commercial, institutional, and industrial sources may be considered as MSW; however, inherently, it does not include construction waste, automobile bodies, municipal sludges, combustion ashes, and industrial process waste even though waste might also be disposed to municipal landfills. MSW compromised a tiny fraction of all the generated waste, but requires most attention due to its direct health impact. Historically, solid waste management was the most rudimentary sort even in the most developed countries. Food wastes usually ended up at a local dump, where open burning was the most common practices to sort out the problems. For comparison, the percentages of waste generated that is landfilled, incinerated, or treated by other means are illustrated in Fig. 8 for different countries. Countries with sparse population such as Australia and Canada tend to landfill most of their generated MSW, whereas densely populated countries such as Japan incinerate more. In present circumstances, scientific waste management in India is a major problematic issue and about 90% of waste is currently disposed of by open dumping or landfilling. However, landfills have also been widely unsuccessful in India because of limited site availabilities. The population of the developing countries is another factor that detrimentally impacts the function of landfill sites. As the population keeps increasing, the garbage quantity also increases, which, in turn, exhausts the landfill sites. Most recently, due to enhancement in consensus of environmental conservation in general people, the technologies to manage the waste have become more sophisticated. Gradual changes in the nature of wastes trigger the concern for environmental conservation and also stimulate the necessity for resource recovery. Recently, scientific landfills have replaced the open dumps, incineration technologies have also been markedly improved with enhanced energy efficiency and pollution control devices, and recycling of waste also become more economical. In order to determine the quantities of MSW generated in a community, there are few different processes, *viz*. input analysis, secondary data analysis, and output analysis. Among these, the output analysis, which is the direct measure of the quantity of the amount of MSW by weighting the refuse dumped at a disposal site, is mostly preferable due to its direct quantification.

Solid waste generation in Indian cities has increased from 6 MT in 1947 to 48 MT in 1997 and is expected to reach 300 MT per annum by 2050. According to National Environmental Engineering Research Institute (NEERI, Nagpur, India) and Central Pollution Control Board (CPCB, India), the per capita waste generation in India varied from 0.17 to 0.76 kg/day. However, absence of formalized collection and waste segregation practices in most of municipalities, coupled with outdated technologies for waste management causes serious health repercussions. According to the National Institute of Urban Affairs,[17] the average waste collection efficiency in 159 cities in India varied from 66% to 77%. The national average of waste collection is mere 72.5%, far below than that of developed countries. Table 1 reveals that per capita generation of MSW has increased to almost 1.5 times, from 375 to 490 gm/day during last 26 years (1971–1997)[18] and plastics accounts nearly 0.5–5% of it. Waste plastics account 0.7% in households across Indian society in 1971 (TERI, 1998),[18] which further increased to a range of 4–9% in 1996[19] (Table 2).

Table 1. Decadal Growth of Per Capita and Total Urban MSW Generation in India.[3,18]

Year	Per Capita Waste Generation (gm/day)	Total Urban Municipal Waste Generated (MT/yr)
1971	375	14.9
1981	430	25.1
1991	460	43.5
1997	490	48.1
2007	500	55.2

Table 2. Decadal Variation of Average Compositions of MSW in India.[3,41]

Components	1970	1995	2005
Paper	3.17	4.64	6.07
Plastics	0.64	3.22	4.88
Metals	0.66	0.43	0.19
Glass	0.38	1.72	0.34
Biodegradable	45.31	52.80	55.06
Ash and fine earth	40.76	26.82	29.6
Other unsorted	9.08	10.37	3.86

Note: All values are in percent dry weight.

3.3. *Application of IWM for MSW Management*

The IWM concept inherently integrates practical waste management system with combination of waste stream, collection, treatment, and disposal methods. The primary objective is to achieve environmental benefits, economic optimization, and societal acceptability. The IWM effectively reduce the quantity of solid waste either by recycling, incinerating, composting, or ultimate disposal of waste residue in scientific manner. No one individual method of waste management can deal with all the materials in a waste stream in an environmental sustainable way and also with economic proficiency. Inherently, waste management is the amalgamation of many closely related processes that are integrated itself. It is always seemed unrealistic to have a major focus on a specific material only because of their ready recyclability (as aluminum) or due to public interferences (as plastics). Therefore, the IWM (Fig. 9) should be sustainable, market oriented, and multi-material approach. The integrated approach of waste management was first proposed by W.R. Lynn, way back in 1962. It was initially discussed as "viewing the problem as interconnected system of component operations and functions." It was well understood that system analyses coupled with mathematical modeling were necessary to optimize waste management operations. Further, in 1975, Solid Waste Authority of Palm Beach Colony, Florida first develop and implement integrated programs of waste management incorporating waste transportation, processing, recycling, resource recovery, and disposal

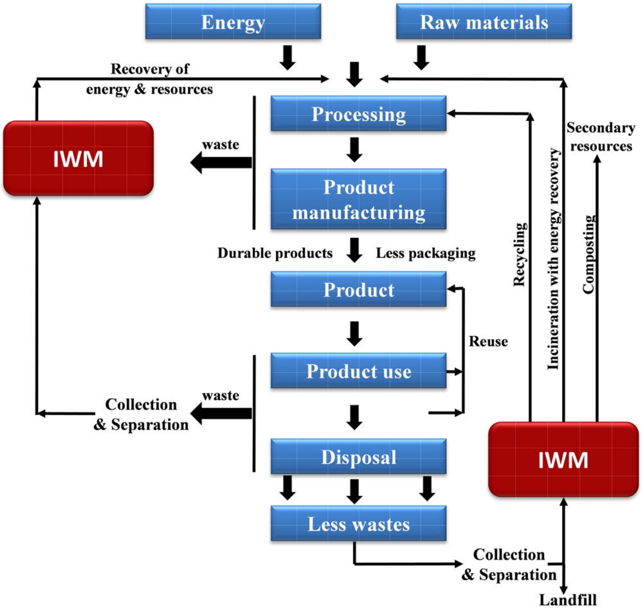

Figure 9. Application of IWM for Waste Management and Maximum Resource Recovery.

technologies. Finally, in 1991, a task force from Economic Commission for Europe defined IWM as a "process of change in which the concept of waste management is gradually broadened to eventually include the necessary control of gaseous, liquid and solid material flow in the human environment." In 1996, United Nations Environment Programme (UNEP) defined IWM as a framework of reference for designing and implementing new waste management and for analyzing and optimizing existing system.

In this resource-depleted world, managing the waste and to fetch maximum resource with minimum economic expenses is now the need that society has to address. Solid waste management practices were initially developed to avoid the adverse health impacts that were being caused by the increasing amounts of waste being generated in the society. In past, solid waste management primarily includes collection, land disposal, and incineration of household waste. Industrial waste disposal did not receive much attention. Even though MSW is dwarfed in size and also in environmental impacts than that of industrial wastes, but generation of industrial waste is only due to virtue of the process of providing society with the materials that ultimately end up in our trash.[5] Therefore, consumption of less, not only saves the wastes that would have ended up in the municipal waste stream, but also reduces the energy, materials, and waste associated with providing those items. Therefore, attention need to be directed not just at the management of consumers waste, but at the complete set of the processes that results in the products our society seems to demand.[5] The Brundtland report clearly established that sustainable development can only be achieved if society in general, and industry in particular, learned to produce "more from less"; more goods and services from less of the world's resources (including energy), while generating less pollution and waste. Therefore, the goal of sustainable solid waste management should be the recovery of more valuable products from the waste with the use of less energy and simultaneously managing more positive environmental impact in terms of human health and safety. In addition to these, a sustainable solid waste management must inherently be more economically affordable and socially acceptable.

The decision regarding waste management should be flexible in terms of its selection from different elements of waste management options, which will result in minimum energy use, environmental impact, and landfill space at a cost affordable to the community. This goal can be achieved by various means such as segregating waste type, recycling of certain types of waste, and beneficial reuse of industrial by-products. The inherent concepts of IWM can also be applied in a community levels essentially consists of the following five steps:

1. Characterization of waste and source identification.
2. Efficient and scientific waste collection.

3. Reduction of volume and toxicity of the waste with adequate treatments.
4. Selection of appropriate technologies in regard to different characteristics of the waste.
5. Optimization of the first four steps to reduce cost and environmental impact and enhance social acceptability.

Solid waste management all too often focuses almost entirely on what to do with a given waste stream, with the key decision being whether to incinerate the waste or bury it. The main goal is to reduce the volume and toxicity of waste being disposed. According some beliefs, recycling is an additional financial burden on a community, whereas land disposal based solely on the current tipping fee, which in most cases is lower than recycling cost. However, landfills do possess limited space and often it closed as unable to meet the new environmental regulations and also the total cost of sitting future landfills is sometimes overlooked. Again in some instances, incinerators were shut down due to its poor performances and as communities became more agitated by the possible environmental effects of incineration. Recycling also possess some constraints as entire municipal waste stream cannot be reduced to zero through a waste recycling program. Therefore, using energy and material balance approach at every stage in life cycle of a product can provide new insights into not only the solid waste problem, but also the problems of air and water pollution.[5]

4. Management of Waste Plastics: Application of IWM Concepts

4.1. *Source Reduction*

Waste minimization, waste reduction, or source reduction is often placed at the top of hierarchy of conventional waste treatment process. Source reduction can be a frontend waste management approach by designing and manufacturing products and packaging with minimum volume and toxic content so as to help ensure that the product has a longer useful life. For the individual consumer or household, source reduction means consuming and throwing away less. Source reduction signifies to any changes in the technology, raw material, designing process, packaging, and the use of materials or products in certain ways to reduce their amount or toxicity before ultimate disposal. Its often act as a precursor to effective waste management, rather than a part of it. Source reduction includes the reuse of material, which is always a better managemental process, because it avoids the costs of waste management. It can be as simple as declining an unnecessary bag for a small purchase or as elaborate as establishing a backyard composting program or choosing cleaning products that do not contain hazardous chemicals.

4.2. *Less Packaging*

Plastics used in packaging represent the largest single sector to contribute in the MSW, like in Germany, it constitutes a 30% share of MSW by weight and 50% by volume.[20] Packaging accounts for 35% in UK, 11% in Germany, and 28% in the USA's total plastics consumption. It is also most important application for plastics in India representing 42% of total plastics consumption.[12] Plastics used in packaging sector are more prone to be dispose off and, therefore, produce environmental degradation. Encouraging society to use less plastics bags and promoting manufacturers to use less plastic in packaging sector and by enhancing its potentiality to be reused, can reduce plastics solid waste up to a large extent.

4.3. *Product Reuse*

The reuse of materials involves either the voluntary continued use of a product for a purpose for which it may not have been originally intended, such as the reuse of a cardboard for a different purpose or the extended use of a product. Reuse is a waste reduction strategy where a product is used for the same or new purpose without undergoing a physical change. Reusing, when possible, is preferable because the item does not need to be reprocessed before it can be used again. Reusable plastics are used more than once and it competes with disposable or single-use products. Reusing plastics is preferable to recycling, as it uses no energy. More durable and multi-trip plastic packaging has become more widespread fashion in India in recent years, replacing less durable and single-trip alternatives.[3]

4.4. *More Durable Product*

Within the principles of waste prevention, the extension of product life cycle is very important aspect. The product that has the more durability certainly reduces the possibility to throw away quickly and, therefore, reduces the quantity of waste generation. Extending the product durability not only reduce waste generation, but also consume less resources for manufacturing the product and, therefore, such kind of practices should be developed in order to enhance product sustainability. Plastic products are much more prone to easy disposal due their low cost, easy availability, and product type. The service life of plastics ranges from 1 to 30 years. However, mostly, the house-hold plastics have the service life much lesser than it should be. Although the plastic compound possess the required durability, still it is our habit that we used to through it away. However, change in public conceptions and practices may reduce the problem. Moreover, through these means, the total service period of plastic compounds is increased and the weighted average service life of plastic compounds helps it to be consumer's first choice.

5. Plastics: IWM & Resource Recovery

5.1. *Recycling of Waste Plastics*

Recycling is the combination of several technologies that are carried out on waste plastics to produce secondary raw materials. Although recycling has a long-lasting history, it has only recently that environmental concerns and waste management issues are considered due to gradual increase in public awareness. Nearly, all the waste materials found in a common MSW are well capable to undergo recycling process with a variable extent of efficiencies. However, the process should be environmentally sound, technically feasible, and economically profitable. Recovery of secondary raw materials through recycling and composting is given the highest priority in the solid management hierarchy after source reduction and reuse. Recycling is somewhat different from reuse, where the materials do not return for remanufacturing. The recycling process indeed requires the participation of public and, therefore, the public must perform the separation of the waste material at initial stage. Due to plastics chemical properties, the recycling of waste plastics possess some technical complications but efficient collection and separation of waste plastics leads enhanced recycling efficiencies.

5.1.1. *Recycling: Comparison of Technologies*

Plastics encompass a wide variety of resins or polymers with different chemical characteristics. In general, plastics fall into one of the two main groups: thermoplastics and thermoset plastics. Roughly, 80% of used plastics are thermoplastics that can be repeatedly formed to a new product by the application of heat. The majorities of house hold plastics comprise polyolefins (polyethylene terepthalate (PET), LDPE, HDPE, or PP), which are thermoplastics and, therefore, are easily recyclable. Polyolefins are a major type of plastic used throughout the world in applications such as soft-drink bottles, clear film for packaging (PET), packaging, bags, containers, pipes (LDPE), milk and water bottles, housewares, industrial wrappings and film (HDPE), automotive parts, film, battery cases, drinking straws, and electrical components (PP) (Table 3). In India, polyolefins share the highest percentage of total plastic consumption followed by PVC and PS, in contrast to 60% of the total thermoplastics in Western Europe. Therefore, the possibility of waste plastic recycling can be more adequate, as it is anticipated that most of the waste plastics will be thermoplasts and hence recyclable. In contrast, the thermoset plastics lose its ability to be reformed or remolded when subjected to heat or pressure and converted to a hard and rigid compound. Addition polymers such as PE cannot be easily recycled by simple chemical methods as in case of condensation polymers. Therefore, thermochemical recycling techniques such as pyrolysis are employed in order to produce

Table 3. Type of Plastic Resins, Recycling Potentials, and Use of Recycled Plastics.

Polymer	Primary Application	Recovery (%)	Recycling Status	Use of Recycled Plastics
Polyethylene terepthalate (PET)	Soft drink and mineral water bottles, textile fibers, processed meat packages, peanut butter jars, pillow, and sleeping bag filling	26	Frequently recycled	Multi-layer detergent bottles, soft drink bottles, and packaging
High-density polyethylene (HDPE)	Milk, water, juice, cream bottles, and shopping bottles	6	Often recycled	Crates, detergent bottles, irrigation pipes, and buckets
Low-density polyethylene (LDPE)	Shopping and garbage bags, cups, and black plastic sheets	0.1	Rarely recycled	Packaging, sheets for nursery, and film for industry
Polyvinyl chloride (PVC)	Automobile seat covers, bottles, shoe soles, electricity pipes, cooking oil bottles, food wrap materials, and building materials		Rarely recycled	Floor materials and covering materials
Polypropylene (PP)	Snack food wrap, straws, car batteries, drinking straws, disposable syringes, medicine bottles, and car seats, batteries, and bumpers	6	Occasionally recycled	Buckets and worm factories
Polystyrene (PS)	Pharmaceutical bottles, disposable cups, packaging materials, laboratory ware, and certain electronic uses	0.1	Rarely recycled	Office accessories, spools, and CD boxes

a series of refined petrochemical products similar with that of commercial gasoline. The concept of recycling may broadly defined as the recovery of material from waste for a purpose that would otherwise require the consumption of virgin resources.[21] Under this concept, plastic recycling process does include both pyrolysis as chemical recycling and incineration for waste to energy recovery. However, here, the incineration of waste plastics is discussed separately from that of recycling in order to have a comparison between two strategies. Although, the concepts of ISWM suggests the judicious application of both technologies in order to achieve the highest material and energy recovery, but here discussion were made separately. However, it is now well established that in order to recover the optimum resources and energy from waste plastics, the recycling process including both chemical recycling and incineration exhibit the highest efficiency and low-environmental repercussions. The various approaches that have been proposed for recycling of waste plastics mainly include:

- Primary recycling,
- Mechanical recycling, and
- Plastic recycling.

Primary recycling is the in-plant process of recycling of waste scraps materials.

Mechanical recycling involves the separation of plastic polymer from its associated contaminants and further reprocessed through melting, shredding, and other related processes. During mechanical recycling of plastic compounds the most important aspect is the separation of different types of plastic resins according to their chemical characteristics. Due to variation in melting points, at a definite temperature, a batch of plastic resins may fully transformed and a different batch may exhibit partial transformations. Therefore, the mechanical recycling of waste plastics are mostly carried out with a single-polymer waste stream and in order to achieve maximum efficiency and homogenous mechanical property of produced goods. Moreover, mechanical recycling mostly operated at a temperature of 200–300°C, which also results in the generation of various toxic gases.

The third type of plastic recycling process is chemical recycling or feedstock recycling, which ultimately leads to complete or partial depolymerization of plastics. Chemical or feedstock recycling also includes pyrolysis, hydrogenation, and gasification. Depending upon the need of secondary materials and the availability of technology coupled with economic feasibility, any one method can be adopted for recycling of waste plastic streams. Chemical recycling is mostly the complete depolymerization of the associated monomers or can be partial degradation of in order to produce secondary commercial products. Of the many alternative chemical recycling processes, pyrolysis has received the most attention. Due to thermal instability of organic compounds, pyrolysis in an oxygen devoid condition results in the combination of thermal cracking and condenzation reactions that ultimately generate various gaseous, liquids, and solids fractions. Pyrolysis is often called as destructive distillation, as it is an endothermic process in contrast to most combustion processes that are exothermic. The major compounds that are generated through pyrolysis depends on the organic characteristics of the compounds but mostly comprises of a gas stream primarily with hydrogen, methane, carbon monoxide, and carbon dioxide, a liquid fraction that consists of a tar oil stream containing mostly acetic acid, acetone, and methanol and a char consisting almost pure carbon with some inert materials. The effective temperature of pyrolysis for waste plastic streams varied from 400–650°C or higher. However, the processes are basically divided into three classes as low, medium, and high temperature based on the required temperature for complete or partial degradation of plastics. Products generated through chemical recycling of plastics mostly depends on feed plastics, feeding rate, effective temperature, residence time, and ultimately on efficiency of the reactor itself. However, the generation of liquid compound enhanced at low temperature in contrast to gaseous compounds at elevated temperature. According to the Achilias *et al.*,[8] with defined catalysts, oil and gaseous fractions were mostly recovered after chemical recycling of HDPE, LDPE, and PP. Generated gaseous and oil fractions have the potential to be reused as a feedstock in the petrochemical industries. Chemical recycling have also been proved as an effective technology to produce fuel as effective as commercial gasoline.[22] Low temperature (400–500°C) chemical recycling of PE and PP mostly results in the generation of high-calorific value gases and waxes coupled with hydrocarbon mixture.[8,22] The produced gaseous fractions possess high calorific value and therefore, can be reused as feedstock and liquid fractions mainly consist of linear olefin and paraffin mixture. The PP and PE at higher temperature ($>$700°C) produce olefin mixture that has the potential to be reused for the production of corresponding polyolefins.[22,23] However, thermal cracking of HDPE and LDPE are somewhat difficult due their low-thermal conductivity.[24] Therefore, for these kinds of plastic resins, chemical recycling at low temperature with effective catalysts are found more appropriate due to reduced energy consumptions coupled with possible formation of specific hydrocarbon compounds.[8,25]

In India, an approximately 4–5% of MSW are post-consumer plastics in comparison to 6–10% in the USA, Europe and other developed countries. Daily generation of post-consumer plastics varied from 4 to 5 thousand tons in India. The percentage of plastic recycling in India is much higher than many of the developed countries in the world. India recycled 47% of total plastic waste in contrast to China (11%), the USA (3.5%), South Africa (15%), and UK (7%). India's average waste plastic recycling rate is also much higher than global average of 20%. There are nearly 20 thousand industries in India in plastics recycling process with total daily capacity of 1 KT. Although plastic recycling in India is mostly mechanized, but generally operates at a low level of organization. There were over 2000 recycling units in India along with 2,500 pelletizers. Easy availability of low-cost manpower and exclusive market for the lower-quality products have helped in flourishing those recycle industries in India.

5.1.2. *Recycling: Technical Complications*

Recycling of waste plastics cannot by itself solve the environmental concerns mainly because of its continuous rise in proportion in MSW and also due to technical complications arises during separation or processing of different types of plastic resins.

The incompatibility between different types of plastic polymers during the recycling is the major issue to be addressed. As example, the presence of a single PVC bottle in a batch of PET type of plastic resins may spoil the whole batch and also have potential to damage the equipment. Different melting point of various plastic components can induce degradation and may develop poor mechanical characteristics. Moreover, the toxic emissions may have significant impact on health and quality of life. Economic consideration of recycling waste plastics should also be addressed in Indian scenario. In order to compute economic efficiencies of recycling process, we should count the costs of collection, material processing, market value of recycled products, and reduced cost saved due to avoidance of landfill. Although in India, recycled plastics have the good market value but public awareness is minimum on health hazard associated with recycling process. Moreover, recycling is not always the best waste management option, due to plastic's inherent properties. During recycling, plastic can only be reprocessed into inferior types of plastics and, therefore, it limits the times it undergoes recycling. Plastics cannot be recycled forever and that is why after being recycled for three to four times, they become unfit for further reprocessing. Plastics recycling process is further complicated by the potential for contamination by products that they once contained.[5]

However, considering the limited available resources of the world, waste prevention and resource recovery should be the fundamental principle to sustainable development. The cost incurred at recycling may be effectively reduced if costs of waste collection and segregation are performed scientifically. In addition, on this part of waste management, public participation can increase the efficiency of the whole system to a great extent. Without the active participation of citizen, a recovery and waste prevention-centric approach cannot be functional in order to effective waste management. In India, fundamental problems for effective waste management are the waste collection, separation, and transportation to the treatment facilities. Poor performances in this aspect lead to fallout of the whole waste management system. The lacks of involvement of public with unawareness of possible health impacts make huge repercussions on the entire waste management system.

5.2. *Plastic Waste to Energy: Incineration*

The most effective way to reduce the volume of solid waste is to burn it in a properly designed and operating condition, the process called as incineration. In an ideal incineration process, the hydrocarbon compounds of the combustible residue combine chemically with the molecular oxygen to generates carbon dioxide and water, and as a residue generates oxides of metals and minerals. The basic advantages of municipal incinerator are that they require less land and also be effectively used for energy generation. Apart from these, the incineration of waste materials has a number of favorable attributes, including volume reduction, immediate disposal without waiting for slow biodegradation process, less land requirement, destruction of hazardous materials, and value addition to waste product by energy recovery or by generating electricity.[5]

5.2.1. *MSW Incineration: Comparison of Technologies*

There are several technologies available for incineration of MSW, leading to different strategies based on the conversion plant itself and on the possible inclusion of waste pretreatment units. Incineration for energy recovery is typically done by two processes. The collected MSW may directly used to mass-burn incinerator or it may be preprocessed to produce a more homogenous product called as refuse-derived fuel (RDF). The RDF mostly comprises of combustible portion of MSW and generally produced through shredding and sorting a relatively uniform segment of MSW. From the energy recovery perspective, MSW incineration through RDF gasification demonstrates maximum efficiency, but it has some negative environmental repercussions. In order to achieve environmental sustainability, a dedicated incinerator with air pollution control devices provides the optimum results. However, in terms of economic profitability, gasification allows the highest revenues from the sale of energy.[26] Selection of an appropriate technology for an MSW incineration system depends critically upon the size of facility under consideration. Several researchers have compared different waste to energy strategies of waste management options. According to Klein,[27] gasification reveals the best electrical conversion efficiency but also posses higher operating costs. Murphy and McKeogh[28] concluded that gasification is more suitable than traditional combustion process. Gasification produces a greater quantity of electricity and results lower specific total costs coupled with lower emissions of CO_2. Baggio *et al.*[29] compared traditional combustion techniques in a dedicated grate combustor with the thermo-select gasification process and the high-temperature pyrolysis process associated with a plasma system. Gasification and pyrolysis technologies generate lesser amount of gas to be treated than traditional combustion. Again, emission factors for most of the gases from traditional combustion are higher than those from gasification, except oxides of nitrogen and mercury. However, if comparisons are made between advanced thermal conversions, such as pyrolysis and gasification, and mass burn combustion technologies then advanced thermal conversion technologies have negligible variations in costs, lower environmental emissions, and higher energy recovery efficiency.[30]

5.2.2. *Incineration: Technical Complications*

However, waste-to-energy faces societal resistance because of community fears that its emission may cause adverse health impacts. Moreover, the incinerator plant is relatively expensive in capital cost as well as operating cost. Again MSW in particular locality may not suit for incineration if it does contain a majority of biodegradable compounds. For instance, MSW high in plastics are very suitable for incineration but unsuitable for biological degradation. In India, incineration is not a common waste management practice, as the garbage is low in calorific value and waste volumes are generally low for a central facility. The MSW composition is largely organic with high moisture content, making it further unsuitable for incineration. The MSW generated in Indian cities mostly comprises of 50% of moisture and nearly 25% of inorganic content, which results a low energy value of 3,000–4,500 kJ/kg. Due to similar reason, an incinerator once installed in Timarpur, Delhi had to shut down in the 1980s. Moreover, the technology for incineration is not available domestically and the import options are highly capital intensive. The performances of various incinerators installed in various cities in India also demonstrate poor performances. The possibility of production of RDF has also been not useful due to nonavailability of a local market. In summary, developing countries such as India do not find this method to be very favorable, especially considering the waste content and the high costs of setting up and running the plants.

Incineration of MSW also leads to generation of toxic emissions that contain heavy metals, dioxins, and other volatile organic compounds (VOCs). Metals *viz.* zinc, cadmium, arsenic, lead, and mercury are the parts of waste stream and, therefore, when incinerated, they become part of generated gases and also persist with soot particles and generated ash. The removal of these metals are simplified when they condensed and absorbed to fly ash particles in contrast to its removal during combustion stage. Ashes produced through mass burn incinerator are also a potential threat of contamination. Roughly 90% of the produced ash in an MSW incinerator is bottom ash that is potentially less toxic than that of fly ash. Mostly, the fly ash is considered as hazardous and, therefore, requires special attention for further treatment. Apart from ash, dioxins, and furans may also produced through MSW incinerators that can be carried long distances from their emission sources, persist for decades in the environment without breaking down into less harmful compounds, and accumulate in soil, water, and food sources. Further, landfills are still required for the disposal of the ash, which adds to the operational cost of an incinerator. A proper incinerator location enhances acceptance, results in economic waste collections, facilities efficiencies, and promotes economic feasibility. The factors that should be considered for site selection of MSW incinerators are:

1. Constant availability of MSW and its characteristic compositions.
2. Isolation with respect to residential area.
3. Direction and persistence of wind and atmospheric stability.

Table 4. Basic Comparison of Heating Values of Different Components of MSW.

	Heating values		
Materials	LHV (MJ/kg)	DHV (MJ/kg)	HHV (MJ/kg)
Paper	15.7	17.4	18.5
Plastics	32.8	33.1	36.5
PE	43.2	43.2	43.7
PS	37.4	37.4	37.5
PU	26.7	26.7	27.8
PVC	23.0	23.1	23.4
Textiles	18.5	17.3	22.5
Food wastes	5.2	11.7	15.2
Glass	0.2	0.2	0.1

Note: HHV: higher heat value or gross heat value, LHV: lower heat value, and DHV: dry heat value.

4. Proper handling of residue and its effective disposal site.
5. Availability of power, skilled people, hygienic and safe transportation, and water.

The energy content of MSW depends on its composition as well as its moisture content. Plastics have high calorific value (36,500 kJ/kg) (Table 4) and it is very much comparable with the coal (28,500 kJ/kg) and fossil fuel (42,500 kJ/kg). However, in polymers where the carbon-hydrogen content is diluted with the presence of other elements such as chlorine, nitrogen, or oxygen, the calorific value gets lower. If waste plastics are separated and processed for energy recovery with efficient emission control then it can be better option to manage waste plastics. On an average, the production of plastic uses 5% of the world's oil as feedstock compared to 85% used for heating and transport. Most of this is recoverable via energy recovery in the form of heat, which can then be converted to electricity.

5.2.3. *Fate of Plastics in Incineration*

To determine the anticipated pollutants from incineration of MSW, the composition of it should be established. Apart from generating air pollutants, other major consideration is the charge that plastics provide to the active gases during the incineration, which ultimately causes corrosion of the equipment. PVC is the major type of plastics that responsible for the generation of hydrogen chloride (HCl). The burning of PVC in incineration temperature above 300°C results in the generation of HCl, water vapor, and carbon dioxide. The residues of incineration are mostly basic in nature and, therefore, neutralize a portion of generated HCl. Additional application of lime to refuse also contributes effectively for the reduction of corrosion due to HCl. Plastics undergo different chemical transformation according to their chemical nature during incineration; therefore, it is important to determine emission potential of it. In incinerator, the degradation of plastics takes place at 100–300°C. The incineration of PVC at 170–300°C generates dioxin, poly aromatic hydrocarbons (PAHs), furan, and HCl.[21] The burning of PVC at 600°C yields a series of aliphatic and aromatic hydrocarbons, among them benzene is the major organic degradation product.[31] According to Baum & Parker,[21] pollutants emission from the incineration of plastics are much higher at lower temperature due to incomplete burning of plastics. Generally, mass-burn incinerator works at a temperature above 700°C and plastics goes complete oxidation at a range of 700–900°C. The generation of dioxin and furans can be reduced to nondeductible levels at temperature over 900°C with residence times over 1–2 s.[5] However, if during incineration dioxin-furan precursors have not been completely destroyed, then as the exiting gases cool, it can react in the presence of fly ash to form new dioxin and furans.

5.2.4. *Comparison between Recycling and Incineration*

Incinerator is a high-temperature heating system and a valuable means of waste disposal. However, there can be significant impacts associated with the plastic incineration process if adequate air pollution control devices are not installed. Again, the process also may not be found feasible in a situation where mostly MSW comprises of biodegradable compounds. Recycling is the means of conserving energy by replacing raw materials in manufacturing products, thereby reducing acquisition of virgin materials from the natural environment. It may be now well demonstrated that economically feasible technologies are available to mechanically sort out the various components of the MSW and, therefore, to implement adequate technology for each components. However, the objectives of the waste prevention or IWM cannot only be achieved through incineration or recycling as both systems may be most effectively only be used as complementary to each other. Some plastics are recycled easily and, therefore, can be recycled to produce a new product. However, recycling a plastic compound restricts its future use in certain sectors and at present situation the market value of recycled components is not encouraging. Substances such as pesticides and oils generally migrate slowly into the plastics and may remain into it after melting or reforming process during recycling. Again recycling by itself will not eliminate the solid waste problems. There are always considerable amount of nonrecyclable plastics residue left for ultimate disposal and, therefore, in this particular situation, incineration may be an encouraging practices. The primary objectives of IWM were waste prevention and resource recovery from the waste materials. Crude oil is mostly used for the manufacture of plastic additives and as a nonrenewable source of energy, the future consumption of this limited resources must be minimized. Recycling waste plastics conserves nonrenewable fuels almost exclusively, but it depends on the chemical nature of plastics itself. Moreover, while the environmental and resource value of the plastic recycling is well established, the economic constrains under current scenario and regulatory conditions are uncertain. Therefore, it will be inadequate to compare the effectiveness recycling and incineration process for solid plastic waste management but simultaneous application of both technologies depending on the characteristics of the plastic material and economic efficiency for a particular situation should be more viable.

5.3. *Landfilling of Waste Plastics*

Landfilling is the means of disposing waste under the soil cover. Since plastics are mostly act as inert materials, therefore, landfill methods are thought to be an effective method for the disposal of waste plastics. Despite the continuous growth of recycling, source reduction and energy recovery, some proportion of the waste will always need to dispose. Further, the simplicity to practice landfilling makes it the most common method for disposing of MSW. However, mostly plastics are resistant against microbial attack and, therefore, remain persistent in environment that results a significant source of environmental pollution, potentially harming life.

In landfill, the degradation of plastics is the process of physical or chemical change in plastic polymer due to several environmental factors, *viz.* sunlight, moisture, temperature, biological activity, etc. The biological degradation of plastic polymers can be defined as the processes that induce changes in polymer properties due to effects of biologically induced chemical and physical reactions that subsequently result the chemical transformations. The principle mechanism for the biodegradation of high molecular weight plastic polymer is enzymatic oxidation or hydrolysis that creates certain functional groups that gradually improves its hydrophilicity. These functional groups make the plastic resins more prone to gradual degradation to its monomer through the enzymatic actions of several microbes. During degradation, the polymer is first converted to its monomers, which further mineralized through the action of variety of microorganisms. Moreover, due to depolymerization of long chain polymer, produced low molecular weight makes it more accessible for further microbial assimilation.[1] However, the extent of biodegradation and the generation of final products depend on several environmental factors. Under aerobic conditions, CO_2 and H_2O are the final products with microbial biomass. In contrast, under anoxic conditions like in landfills or composting, primary products are the microbial biomass, CO_2, CH_4, and H_2O.[32] However, complete biodegradation of plastic polymers can hardly be achieved as the portion of polymer will generally incorporated into microbial biomass or any other natural products. Degradation of PE in landfills has been reported by different researchers through various mechanisms such as chemical degradation, thermal degradation, photodegradation, and biodegradation. Mostly, PE is prone to photodegradation and/or chemical degradation, which is mainly achieved through two principle mechanisms: hydrobiodegradation and oxobiodegradation.[33] However, some researchers also reported that PE sheet incubated in moist soil for 12 years shows no signs of deterioration[34] and only partial degradation could be achieved after 32 years.[35] Biodegradation of PS has not been extensively reported, but microbial decomposition to its monomer has been described by Tsuchii *et al.*[36] The PVC can be subjected to thermal degradation and photodegradation but biodegradation of PVC is only achieved partially by the application of white-rot fungi.[37]

However, traditional plastics are not biodegradable and, therefore, only disintegrate into tiny particles that eventually liberate different compounds that have potential to contaminate groundwater, if not properly managed. Moreover, burying waste plastics in landfill is not a proper IWM approach, as it only reduces any possibilities for recovery of any secondary raw materials or energy for further use. Traditional plastics production requires large amounts of natural resources, primarily nonrenewable fossil fuels, both as a raw material and to provide energy for the manufacturing process. Roughly, 10% of the world's annual petroleum production is consumed in the production of plastic resins; 4% is used as a raw material and an additional 6% is consumed in the form of fuel or energy during manufacture. Eventually, if the waste plastics are landfilled, then it reduces any possibilities to recover the secondary raw materials from these waste products. Generation of waste not only has significant environmental impacts, but it also places a great pressure on the finite resources of the earth. Therefore, waste management decision must be tied inextricably to conserve resources and its utilization. Moreover, burying the plastics waste blocks the natural supply of air and water to the soil, thereby affecting plant life, reducing the water-retaining capacity of the soil and affects the water table.

5.4. *Plastic Waste Management: Recent Approaches*

The productive use of waste material represents a means of alleviating some of the problems of solid waste management. Since, the disposal of waste plastic in landfill has several harmful effects on the environment; therefore, the logical methods for reduction negative effects are the applications of these materials in other industries. The application of waste plastics as a fuel in cement kilns has a potential to be an effective measure of waste reduction. Moreover, due to extreme temperature inside kilns, the possibility of generation of any toxic gases also reduces. Waste plastics are most suited to be use as a fuel as it has a calorific value well comparable to that of conventional fuel. However, chlorinated compound as PVC is especially considered because produced HCL and chlorine gas can be readily neutralized. The efficient use of waste plastics as a fuel has further environmental benefits, as no solid or ash residues are produced and air emissions are not greater than fossil fuels. A cement plant of annual 1 MT capacity have the potential to consume about 10–30 thousand MT of plastics waste, which reveals the actual potential of this technology to be most effective to sort out waste plastic disposal problems. In recent years, large fractions of energy needed for the production of cement comes from the waste used as an alternative fuels. Plastic wastes play a significant contribution to the replacement of nonrenewable fossil fuels such as coal or petroleum. In several European countries, there is already growing competition between the cement industries and MSW incinerators producing energy, both competing for the use of plastic waste as an alternative fuels. However, the use of processed plastics as fuel for cement kilns is often considered as in term of energy recovery.

The waste plastics have the potential to be reused as an alternative fuel in the blast furnace and often considered as recycling of waste. For the smelting of iron ore (FeO), traditionally, coal is used to fuel the furnace and to generate carbon monoxide (CO) and heat. This reaction occurs when CO gas, released from the burning coal, combines with the oxygen molecule from iron oxide to produce iron and carbon dioxide (CO_2). Recently, waste plastics have replaced substantial portion of coke or pulverized coal for producing pig iron from iron ore. Waste plastics burnt in the absence of sufficient oxygen similar to that of coal and produces CO that further used as a reducing agent to convert iron ore to pig iron (Fe). Moreover, extreme temperature (over 1,500°C) inside the blast furnace reduces the ash generation and any possibilities of dioxin and HCl gas formation. However, eventually if any minute amounts of dioxin or HCl do produce at blast furnace may readily neutralize by limestone. However, it is essential to identify the optimal ratio of plastic to coal as when plastic is added, hydrogen molecules from the plastic combine with oxygen to produce water vapor (H_2O), iron and reduced amounts of CO_2. Water reacts more aggressively than CO_2 under high temperatures and causes other fuels in the furnace to degrade. Moreover, the presence of PVC at waste plastic stream produces chlorine gas that has the potential to corrode the furnace. In order to prevent the diffusion of chlorine inside the blast furnace, coal tar and converter dust have excellent implication.[38]

Except from the principles of waste prevention, the recycling of waste plastics has become most promising and environmental friendly among the other means of IWM methodologies. However, the recycling of waste plastics can be achieved by several means depending on the characteristics of the plastic polymer and required product itself and, therefore, can be classified into primary, secondary, tertiary, and quaternary forms. Among these, the most efficient method in terms of environmental conservation is feedstock or tertiary recycling that mainly useful to covert waste polymers to original monomers or other valuable chemicals compounds. Feedstock recycling principally converts waste plastics to their constituent monomers that are then further processed either separately or with petroleum residues to derive conventional fuel. With the presence of catalytic additives, long-chain plastic polymers split to liquid, solid, and gaseous series of refined petrochemical products and particularly of a liquid fraction very similar with that of gasoline.[22] It has been previously discussed that at higher temperature due to thermal cracking of PE and PP, a mixture of olefin with certain aromatic compounds and high-calorific value gases are achieved. Chemical recycling of waste plastics *viz.* HDPE, LDPE, and PP with specific catalysts generate aliphatic compounds that can be easily used as a raw material for the production of refined fuels.[8] However, coprocessing of petroleum residue with waste plastics helped to enhance the hydrogen content of the final product and makes it more useful. Therefore, apart from recycling only plastic polymer, researchers have also studied coliquefaction of coal and waste polymers. On a laboratory scale experiment, it was found feasible to convert waste plastics coupled with conventional coal to liquid fuel at a relatively high temperature (400–450°C) and moderate to high hydrogen pressure.[39]

6. Conclusion

In brief, this article emphasizes on the increasing trend of global as well as Indian plastic production and consumption scenario. Moreover, with the concepts of ISWM, plastic waste disposal problems are tried to sort out. Plastics are the integral part of the society due to its extreme versatility and durability, light weight, excellent thermal and electrical insulations, chemical resistance, and safety in regards to its competing materials. However, coupled with all these properties and its relative inexpensiveness have made these plastics much more prone to easy disposal and, therefore, causing concern for environmental safeguard. When plastic products are used and discarded, these plastics and additives are undesirable from an environmental view point. Traditional plastics are not biodegradable and are extremely difficult components for landfilling for its volume and any future possibilities of groundwater and soil contaminations. Incineration is generally not found technically feasible in most of the developing countries and also possess chances of air emission if not scientifically managed. Recycling of waste plastics is the most attractive method in accordance with the principles of sustainable development but can only be achieved for a limited period as only inferior type of plastic can be produced through recycling with several use restrictions.

All these emphasize that no one individual method is sufficient to deal effectively with all materials in waste in an environmental sustainable way. In reality, if we consider IWM principles, then it emphasizes that any waste management is built up of many closely related and integrated processes. Therefore, instead of focusing on and comparing individual options, attempt should be made to integrate waste management systems in such a way that it can deal with the whole waste stream, and then compare their overall performances in environmental and economic terms. In fact, an IWM system can itself become a part of a resource management system, where all resources are managed within a single optimized system.

However, in order to reduce the waste plastic problem what we can do is to adopt the principles of waste prevention. Waste prevention undoubtedly needs to be a central theme of social responsibility. Apart from this, considerable shifts in perception and behavior among consumers are also essential in order to either restrict or to minimize the use of plastics. The use of plastics need not be reduced but what is of urgent required is to use the plastic compound judiciously and to promote reuse of plastics. If plastic compounds are made more durable and if general the perception of consumer is changed regarding the reuse of plastic and less disposal, then inherently the waste plastic problem can be sort out. However, in Indian situations, there are several constraints such as proper collection, segregation, and transportation of the discarded plastic material. However, increase in public awareness coupled with changes in individual behavior can be an effective way to reduce the environmental repercussions of waste plastics. Apart from these, in a resource limited world, the recovery of energy and resources should be fundamental principle to sustainable development and in order to achieve its active public participation and proper implementation of regulations are essential.

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Chapter 18

WASTE SLUDGE MANAGEMENT IN WATER UTILITIES

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Abstract

Surface raw waters contaminated by humic acid (27–405 mg/L) and having extremely high color (425–5750 color units) were successfully treated by magnesium coagulant, lime, and polymer. After coagulation and clarification treatments, over 98% of the color was removed when optimum dosages of magnesium chloride, calcium hydroxide, and polymer Nalco-625 were determined by a streaming current detector and applied to raw waters during proper mixing and flocculation. Both sedimentation (2–24 h. D.T.) and dissolved air flotation (2.5–20 min. D.T.; 33% recycle) are feasible for clarification, resulting a 2.3–3.1% of clarifier sludge. Either sedimentation effluent or flotation effluent had to be treated by recarbonation and sand filtration in order to meet the drinking water standards. The clarifier sludge containing mainly magnesium and calcium was subsequently conditioned with 20–100 mg/L polymer Magnifloc 905N, thickened to 16.7–25% with a sludge reducer, and dewatered to 40–43% with a filter press. The dewatered sludge was incinerated at 600°C for recovery of magnesium and calcium chemicals. The recovered chemicals containing no organics were proved to be good coagulants. Alternatively, the dewatered sludge can be buried in a sanitary landfill operation. Another feasible dewatering technique developed was microwave process for sludge dewatering.

Keywords: Waste sludge, water utility, magnesium coagulant, sedimentation, dissolved air flotation, chemical recycle.

1. Recovery and Reuse of Magnesium Coagulant from a Simulated Water Treatment System that Generates Magnesium Sludge

1.1. *Project Concept of Magnesium Sludge Recovery System*

Water or wastewater containing various pollutants can be treated with magnesium coagulant that precipitated as magnesium hydroxide flocs in the presence of lime. The resulting sludge is composed of $CaCO_3$, $Mg(OH)_2$, and the pollutants that are removed from water or wastewater.

The sludge can be: (a) conditioned, thickened, and dewatered into dry cake for sanitary landfill, (b) carbonated by injecting CO_2 gas to dissolve the $Mg(OH)_2$ for magnesium coagulant recovery, or (c) incinerated for production of MgO and CaO for reuse.

The researchers at the Lenox Institute of Water Technology have prepared humic acid contaminated raw water to investigate the magnesium sludge conditioning, sludge thickening and alum recovery (STAR) System. The experimental results and conclusions are reported in two parts.

Both sedimentation and flotation are well-established water clarification processes. Flotation, sludge reducer, and gravity thickener are effective sludge thickening processes. Flotation, therefore, has the functions of water clarification as well as sludge thickening.

Filter belt press and microwave are two sludge dewatering processes under this investigation. The microwave process equipment, sludge reducer, and filter belt press are fully described elsewhere.[1]

1.2. *Treatment of Magnesium Sludge from Clarification of Water Containing 27 mg/L Humic Acid and 425 Color Units*

1.2.1. *Water Clarification and Sludge Production*

The Lenox raw water contaminated by 27 mg/L of humic acid was analyzed as follows:

pH = 8.1 unit
Color = 425 color unit
Alkalinity = 60 mg/L $CaCO_3$
Total hardness = 100 mg/L $CaCO_3$
Calcium hardness = 80 mg/L $CaCO_3$
Magnesium hardness = 20 mg/L $CaCO_3$
TSS = <1 mg/L
COD = 20 mg/L
Humic acid = 27 mg/L

The Lenox raw wastewater was adjusted to $pH = 11.3$ with 370 mg/L lime and treated with 47 mg/L $MgC1_2$ and 2 mg/L Nalco 625. The well-flocculated wastewater sample was then processed through either conventional sedimentation in an Imhoff cone or innovative flotation for liquid and solids separation. The settleability or floatability can be found from another technical paper.[2]

The separation of liquid and solids by sedimentation was effective. The settled sludge consistency was 2.3% after 2 h of settling.

The separation of liquid and solids by flotation at 33% recycle flow pressurization mode was also excellent. The floated sludge consistency was about 2.4% after 3 min of flotation.

The clarified effluent quality either sedimentation or flotation was the same:

pH = 11.5 unit
Color = <1 (true) color unit
Alkalinity = 300 mg/L $CaCO_3$
Total hardness = 460 mg/L $CaCO_3$
Calcium hardness = 440 mg/L $CaCO_3$
Magnesium hardness = 20 mg/L $CaCO_3$
TSS = 17.5 mg/L
COD = 10 mg/L

The clarified water was further recarbonated with carbon dioxide and filtered by sand (E.S. = 0.35 mm; U.C. = 1.6; 2.5 gpm/ft^2). The filter effluent was analyzed as follows:

pH = 8.2 unit
Color = 0 color unit
Turbidity = 0.4 NTU
Total hardness = 110 mg/L $CaCO_3$
TSS = <1 mg/L

1.2.2. *Sludge Process System Including Polymer Conditioning and Sludge Reducer Screening*

The produced sludge (2.3–2.4% consistency) was conditioned with 20 mg/L Magnifloc 905N, screened with a sludge reducer. The screened sludge solids content was 25%.

This entire process system is concluded to be technically feasible if sanitary landfill is intended.

1.2.3. *Process System Including Polymer Conditioning, Sludge Reducer Screening, and Microwave*

The sludge produced from water clarification (2.3–2.4% consistency) was polymer conditioned and screened through sludge reducer then microwave processed for 1 min. The final solid content was 53%.

Table 1. Lenox Institute of Water Technology Belt Press Laboratory Test Results.

Weight Applied	Dewatered Sludge Cake (%)
22 pounds	43
50 pounds	44
80 pounds	46

Notes: I. Chemical Dosage
Magnifloc 905N = 20 mg/L by weight.
II. Filtrate/Pressing Results
Initial sample: 300 mL volume; NA% total volume; 2.4% dry solids.
Gravity drainage: NA mL volume; NA% total volume; NA % dry solids.
First press drainage: 15 mL volume; 5% total volume; 43% dry solids.
Second press drainage: 5 mL volume; 1.7% total volume; 44% dry solids.
Press cake: 250 mL volume; 83.3% total volume; 46% dry solids.

The sludge process system is concluded to be excellent if a giant microwave process unit is commercially available.

1.2.4. *Sludge Process System Including Polymer Conditioning, Sludge Reducer Screening, and Belt Filter Press*

The 2.4–2.5% of raw sludge produced from water clarification (by sedimentation or flotation) was polymer conditioned with 20 mg/L Magnifloc 905N and screened with reducer. The screened sludge sample was dewatered by a belt filter press Type Ekosep-PL simulation (Table 1). The final results were very good. This sludge process system is also technically feasible.

1.3. ***Treatment of Magnesium Sludge from Clarification of Water Containing 405 mg/L Humic Acid and 5750 Color Units***

1.3.1. *Water Clarification and Sludge Production*

The Lenox raw water contaminated with 405 mg/L of humic acid was analyzed as follows:

pH = 8.0 unit
Color = 5750 color unit
Alkalinity = 60 mg/L as $CaCO_3$
Total hardness = 70 mg/L as $CaCO_3$

Calcium hardness = 60 mg/L as $CaCO_3$
Magnesium hardness = 10 mg/L as $CaCO_3$
TSS = 14 mg/L
COD = 380 mg /L
Humic acid = 405 mg/L

The highly contaminated raw water was adjusted to pH 11.5 with 5550 mg/L lime and treated with 700 mg/L $MgC1_2$ and 30 mg/L Nalco 625. The flocculated sample was processed with either sedimentation or flotation for the separation of liquid and solids. The settleability and floatability of the flocculated sample can be found from another report.[2]

The sludge consistency after 24 h settling was approximately 3%. The effluent quality of sedimentation was analyzed as follows:

pH = 10.7 unit
Apparent color = 175 color unit
True color = 20 color unit
Alkalinity = 120 mg/L $CaCO_3$
Total hardness = 560 mg/L as $CaCO_3$
Calcium hardness = 440 mg/L as $CaCO_3$
Magnesium hardness = 120 mg/L as $CaCO_3$
TSS = 76 mg/L
COD = 39 mg/L

The effluent quality of dissolved air flotation (DAF) at 33% recycle flow pressurization mode was:

pH = 10.5 unit
Apparent color = 90 color unit
True color = 20 color unit
Alkalinity = 60 mg/L as $CaCO_3$
Total hardness = 440 mg/L as $CaCO_3$
Calcium hardness = 400 mg/L as $CaCO_3$
Magnesium hardness = 40 mg/L as $CaCO_3$
TSS = 2 mg/L
COD = 38 mg/L

The DAF floated sludge consistency was about 3.1% after 3 min of detention time.

Both the sedimentation-clarified water and the DAF-clarified water were further recarbonated with carbon dioxide and filtered with sand (E.S. = 0.35 mm;

U.C. = 1.6; 2.5 gpm/ft^2). The filter effluent was analyzed as follows:

pH = 8.5 unit
Color = 1 color unit
Turbidity = 0.4 NTU
Total hardness = 120 mg/L as $CaCO_3$
TSS = <1 mg/L

It should be noted that the consistencies of settled sludge (3%; 24 h) and floated sludge (3.1%; 3 min) were not high enough. An optional gravity thickener was used to thicken the magnesium sludge to 6% for cost saving in the subsequent polymer-conditioning process. The gravity thickener, however, can be eliminated in all the sludge process systems studied and discussed below for saving in capital and construction costs. Without a gravity thickener, all the sludge process systems presented below will still function properly except that the chemical cost of polymer conditioning will be higher.

1.3.2. *Sludge Process System Including Gravity Thickening, Polymer Conditioning, and Sludge Reducer Screening*

The concentrated sludge (6.0% consistency) sample was conditioned with 100 mg/L of Magnifloc 905N, screened with a sludge reducer. The screened sludge solids content was 16.7%, which was not high enough for sanitary landfill. This process system is not feasible for proper sludge treatment.

1.3.3. *Sludge Process System Including Gravity Thickening, Polymer Conditioning, Sludge Reducer Screening, and Microwave*

The sludge from a gravity thickener with 6% solids content was polymer conditioned with 100 mg/L Magnifloc 905N and screened with sludge reducer. The screened sludge sample was dewatered by means of microwave method for 1 min. The final solid content was 28%, which was excellent. This sludge process system is feasible if a commercial microwave process unit is readily available.

1.3.4. *Sludge Process System Including Gravity Thickening, Polymer Conditioning, Sludge Reducer Screening, and Belt Filter Press*

The gravity-thickened sludge, with 6% solid content, was polymer conditioned with 100 mg/L Magnifloc 905N and screened with sludge reducer. The screened sludge

Table 2. Lenox Institute of Water Technology Belt Press Laboratory Test Results.

Weight Applied	Dewatered Sludge Cake (%)
22 pounds	40
50 pounds	36
80 pounds	34

Notes: I. Chemical Dosage
Magnifloc 905N = 100 mg/L by weight.
II. Filtrate/Pressing Results
Initial sample: 375 mL volume; NA% total volume; 6% dry solids.
Gravity drainage: NA mL volume; NA% total volume; NA % dry solids.
First press drainage: 10 mL volume; 3.3% total volume; 41% dry solids.
Second press drainage: 10 mL volume; 3.3% total volume; 37% dry solids.
Press cake: 260 mL volume; 86.7% total volume; 35% dry solids.

sample was further dewatered by a belt filter press Type Ekosep-PL simulation (Table 2). The final results were excellent.

This sludge process system is feasible and recommended.

1.3.5. *Sludge Process System Including Gravity Thickening and Recarbonation (A STAR System)*

The gravity-thickened sludge (6% solid content) was bubbled with carbon dioxide in a pressurized reactor at 40 psig for 1 min. The magnesium sludge was easily solublized to form magnesium bicarbonate, a reusable magnesium coagulant. The color of recovered magnesium coagulant, however, was extremely high.

It is concluded that recarbonation is an excellent process for recovery and reuse of magnesium coagulant only under the condition that the organic contents of raw water and magnesium sludge are not high.

1.3.6. *Sludge Process System Including Gravity Thickening, Polymer Conditioning, Sludge Reducer Screening, Belt Filter Press, and Incineration (A STAR System)*

After the clarifier sludge (3%) was thickened (to 6%), conditioned/screened (to 16.7%), and pressed (to 40% at 22 pounds), the product sludge was dry and dark brown in color because of its high organic content.

The dry sludge cake was incinerated at 800–900°C for 60 min for purification. The final product was white dry flakes containing magnesium oxide and calcium oxide.

The end product was ground to powdered form and reused as a coagulant for treatment of a fresh Lenox raw water containing 27 mg/L of humic acid with 425 color units. After flocculation, clarification, recarbonation, and filtration, the final filter effluent had zero color.

It is concluded that if the organic contents of raw water and magnesium sludge are high, the Lenox Institute of Water Technology newly developed STAR system will adequately recover the magnesium coagulant as well as lime for reuse in water purification although for wastewater treatment will be more adequate.

2. Recovery and Reuse of Filter Alum and Soda Alum from a Water Treatment Plant

2.1. *Introduction of STAR System*

The STAR system has been developed for pollution and resource recovery. The STAR system is composed of a DAF unit for sludge thickening, an acid reactor for filter alum recovery, an alkaline reactor for soda alum recovery, and two liquid–solids separators for removal of inert residues from the two reclaimed liquid alums. This section presents the technical data generated from a demonstration project. Both the reclaimed filter alum and reclaimed soda alum are as effective as their virgin equivalents for water purification and phosphate removal.

2.1.1. *Water Purification Systems and Waste Streams*

A conventional water treatment plant consists of flash mixers, flocculators, sedimentation clarifiers, filters, disinfection units and perhaps also corrosion control facilities.[3] Average total detention time of conventional plants is in the range of 7–9 h. Recently, a high-rate innovative water treatment plant with a total detention time of only 15 min has been developed and proven to be feasible in the Town of Lenox and the City of Pittsfield, both in Berkshire County, Massachusetts, the USA.

The innovative water treatment plants are composed of mixing, flocculation chamber, DAF module, filtration module, clear well, and disinfection facilities.

Both conventional and innovative water plants use the same types of coagulants (filter alum, soda alum, ferric sulfate, polymers, etc.) and produce the same types of waste sludge (aluminum hydroxide, ferric hydroxide, etc.).

Filter alum (aluminum sulfate) is the most commonly used coagulant. Soda alum (sodium aluminate) has steadily gained popularity since 1980, and is a coagulant as well as an alkalinity supplement. The end products of both filter alum and soda alum are aluminum hydroxide sludges.

An inorganic polymer, known as "poly aluminum chloride (PAC)," has been proven to be effective for water purification in winter.[4,5] After hydrolysis reaction, PAC is also converted into aluminum hydroxide sludge.

Ferric sulfate is another coagulant feasible for water purification under cold water temperatures. The end product of this coagulant is ferric hydroxide sludge which must be disposed of.

Another inorganic polymer, poly iron chloride (PIC), is as effective as PAC for water purification.[6] If PIC is used for coagulation, the waste sludge produced is also ferric hydroxide.

In summation, the two most common waste sludges generated from water treatment plants are aluminum hydroxide and ferric hydroxide. Although many feasible thickening, dewatering, and ultimate disposal methods have been developed,[7–41] a majority of water treatment plants cannot afford to implement conventional sludge treatment and disposal technologies.

2.1.2. *Handling and Disposal of Water Plant Sludge*

Techniques for the treatment and disposal of water treatment plant's sludge can be divided into three general categories, mechanical thickening/dewatering, nonmechanical treatment, and direct disposal and reuse.

Mechanical alum sludge thickeners include DAF thickeners and gravity (sedimentation) thickeners. The former are technically much more feasible than the later. A DAF thickener can easily thicken the alum sludge from 2500 mg/L to 2.5% with the aid of organic polymer.[28]

At present, only centrifugation, vacuum filtration, and pressure filtration have gained acceptance in water utility industry for alum sludge dewatering. A new sludge-water separation unit developed by Krofta Engineering Corporation and the Lenox Institute of Water Technology for sludge dewatering includes both centrifugation and filtration actions. All mechanical methods are expensive, but they produce a reasonable product that is suitable for either alum recovery or ultimate disposal. Smaller plants usually use nonmechanical thickening and dewatering technologies for handling and disposal of their alum sludge. The common nonmechanical technologies include lagooning, air drying beds, and freezing. In many areas of the country, the local weather conditions make nonmechanical alum sludge thickening and dewatering technologies impractical. Besides, the thickened and dewatered alum sludge by nonmechanical means are poor in quality and cannot be reclaimed for alum recovery.

When possible and acceptable, the alum sludge from a water treatment plant can be directly discharged into a nearby municipal wastewater treatment plant for removal of BOD, COD, TSS, and phosphate.[5,15,17] Pittsfield Water Treatment Plant, which is also a potable flotation plant, successfully discharges its alum sludge to the Pittsfield Wastewater Treatment Plant for BOD and phosphate removals.[5] Pittsfield Wastewater Treatment Plant is a biologically activated sludge plant with anaerobic sludge digesters.

Another acceptable ultimate sludge disposal method is sanitary landfill. Alum sludge recovery and reuse is the most promising approach and is highly recommended.

2.2. *Alum STAR System*

A STAR system has been jointly developed by Krofta Engineering Corporation and the Lenox Institute of Water Technology for wastewater applications[16] and potable water application.[17]

The STAR system includes: sludge thickening by DAF (either a flotation clarifier or a flotation–filtration clarifier), filter alum recovery by acid reactor, soda alum recovery by alkaline (or base) reactor, filter alum purification by a water–solids separator, and soda alum purification by another water–solids separator. The reclaimed filter alum and soda alum solutions are separately returned to the mixer and flocculator for water purification, as needed. Figure 1 shows the flow diagram of a STAR system.

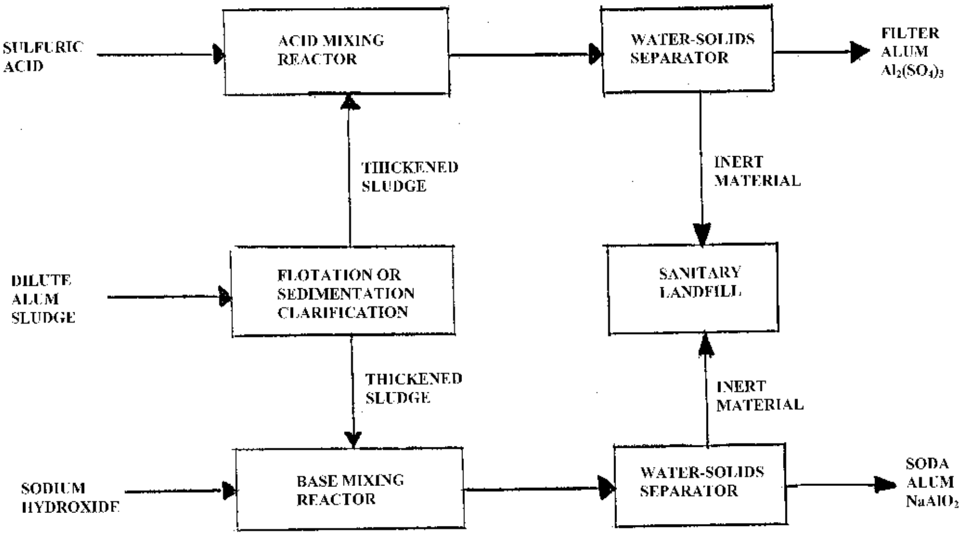

Figure 1. Sludge Thickening and Alum Recovery (STAR) System.

2.3. *Lenox Water Treatment Plant Demonstration Project*

2.3.1. *Water Purification*

The STAR demonstration project was financially sponsored by Krofta Engineering Corporation, Lenox, Massachusetts, the USA, and was conducted by the Lenox Institute of Water Technology.

The demonstration site was the Lenox Water Treatment Plant, Massachusetts[18,19] where there was a flotation–filtration clarifier (Krofta Sandfloat Model SAF-12; diameter = 22 ft = 6.7 m; depth = 6 ft = 1.8 m; detention time = 15 min; design capacity = 1.2 MGD = 4.542×10^6 L per day) for water purification demonstration. Sandfloat is a flotation–filtration package plant consisting of mainly mixing/flocculation, DAF, sand filtration and clear well. It is commercially available from Krofta Engineering Corporation. The Lenox plant has been successfully operated by the Town of Lenox operators since July 1982.[21]

At the time of this investigation, the Lenox flotation–filtration plant was treating 560–770 gpm (2119–2914 L per min) influent flow using 3.9–4.6 mg/L of filter alum and 1.5 mg/L of soda alum (both as $A1_2O_3$). The plant recycled 100% of filter backwash water and produced 3.0–3.6 gpm of floated waste sludge. The following were the average Lenox raw water quality and flotation–filtration effluent quality during this study:

Upper Root Reservoir Raw Water Quality
Temperature = 56°F = 13.3°C
pH = 6.7 units
Color = 25 units
Turbidity = 2.1 NTU
Alkalinity = 44 mg/L as $CaCO_3$
Aluminum = 0.06 mg/L
Iron = 0.02 mg/L
Phosphate = 0 mg/L

Lenox Plant's Flotation–Filtration (Sandfloat) Effluent Quality
pH = 6.85 units
Color = 0 units
Turbidity = 0.3 NTU (during filter backwash)
0.2 NTU (between filter backwash)
Alkalinity = 38 mg/L as $CaCO_3$
Aluminum = 0.05 mg/L
Iron = 0.01 mg/L
Phosphate = 0 mg/L

It can be seen that the flotation–filtration clarifier's performance was excellent, and its effluent turbidity was only 0.3 NTU even when 100% of filter backwash wastewater was recycled.

The fresh raw alum sludge generated from the Lenox flotation–filtration plant was used throughout this investigation. The reclaimed filter alum and soda alum were recycled to the plant for treating the same Lenox raw water.

2.3.2. *Sludge Thickening*

All fresh raw alum sludge at the Lenox Water Treatment Plant was thickened by DAF before acid treatment for filter alum recovery and alkaline treatment for soda alum recovery. The efficiency of DAF for alum sludge thickening has been reported by Krofta and Wang.[17,19] In general, DAF thickens the dilute alum sludge (2500 mg/L) to about 2.5% in consistency.

2.3.3. *Filter Alum Recovery*

Extensive filter alum recovery experiments were conducted at both the Lenox Water Treatment Plant and the Lenox Institute of Water Technology. Results are fully documented elsewhere.[17]

The STAR system operational conditions and the quality of reclaimed filter alum are partially reported below:

STAR Operational Conditions
Dilute raw alum sludge = 2510 mg/L TSS
Alum sludge thickened by DAF = 25830 mg/L TSS
Aluminum content of thickened sludge = 5279 mg/L
Acid = 36-N conc. H_2SO_4
Acid-mixing reactor contact time = 1 h
Acid-mixing reactor mixing rate = 30 rpm
Volumetric ratio of conc. H_2SO_4 vs. thickened sludge = 250/10000
Acid reactor temperature = 22°C
Acid reactor effluent treatment = Centrifugation

Reclaimed Filter Alum Quality
Soluble aluminum = 4660 mg/L
Soluble iron = 136.5 mg/L
THM = 0 mg/L

TOC	= 1528 mg/L
COD	= 1000 mg/L
Total coliforms	= ND (nondetected)
TSS	= 2038 mg/L
VSS	= 1078 mg/L
FSS	= 960 mg/L
Arsenic	= ND
Barium	= ND
Cadmium	= 0.01 mg/L
Chromium	= ND
Copper	= 3.24 mg/L
Lead	= 1.3 mg/L
Manganese	= 8.86 mg/L
Mercury	= ND
Nickel	= 1.5 mg/L
Platinum	= ND
Potassium	= 18 mg/L
Selenium	= ND
Sodium	= 43 mg/L
Zinc	= 0.35 mg/L
Titanium	= ND
pH	= <2 units

2.3.4. *Soda Alum Recovery*

Tables 3–6 fully document the data on soda alum recovery. It appears that the optimum reaction time for soda alum recovery could be 0.25 h (15 min) or shorter. The selected STAR system operational data and the quality data of reclaimed soda

Table 3. Recovery of Sodium Aluminate from Lenox Alum Sludge (Test No. A; TSS = 30914 mg/L) using Ten Normal Sodium Hydroxide.

Reaction Temperature (°C)	Reaction Time (Hour)	mL Sludge Plus mL 10 N NaOH	Recovered Soluble Aluminum (mg/L)	Aluminum Recovery (%)
18	0	10000 + 0	0.1	0
19	0.25	10000 + 1000	5106	86.5
20	0.50	10000 + 1000	4681	79.3
20	1	10000 + 1000	4583	77.7
19	24	10000 + 1000	4276	72.5

Table 4. Recovery of Sodium Aluminate from Lenox Alum Sludge (Test No. B; TSS = 30914 mg/L) Using Ten Normal Sodium Hydroxide.

Reaction Temperature (°C)	Reaction Time (Hour)	mL Sludge Plus mL 10 N NaOH	Recovered Soluble Aluminum (mg/L)	Aluminum Recovery (%)
Test B1				
18	0	10000 + 0	0.1	0
20	0.25	10000 + 700	4866	82.5
20	0.50	10000 + 700	4588	77.8
20	1	10000 + 700	4657	78.9
19	24	10000 + 700	4345	73.6
Test B2				
18	0	10000 + 0	0.1	0
19	0.25	10000 + 500	3427	58.1
20	0.50	10000 + 500	3050	51.7
20	1	10000 + 500	3135	53.1
19	24	10000 + 500	2591	43.9

Table 5. Recovery of Sodium Aluminate from Lenox Alum Sludge (Test No. C; TSS = 30914 mg/L) Using 10 N Sodium Hydroxide.

Reaction Temperature (°C)	Reaction Time (Hour)	mL Sludge Plus mL 10 N NaOH	Recovered Soluble Aluminum (mg/L)	Aluminum Recovery (%)
Test C1				
18	0	10000 + 0	0.1	0
19	0.25	10000 + 400	2473	41.9
20	0.50	10000 + 400	2409	40.8
20	1	10000 + 400	2395	40.6
19	24	10000 + 400	1655	28.1
Test C2				
18	0	10000 + 0	0.1	0
19	0.25	10000 + 300	1571	26.6
20	0.50	10000 + 300	1376	23.3
20	1	10000 + 300	1576	26.7
19	24	10000 + 300	723	12.3

alum are reported below:

STAR Operational Conditions
Dilute raw alum sludge = 2460 mg/L TSS
Alum sludge thickened by DAF = 30914 mg/L TSS
Aluminum content of thickened sludge = 5901 mg/L

Table 6. Recovery of Sodium Aluminate from Lenox Alum Sludge (Test No. D; TSS = 30914 mg/L) Using Ten Normal Sodium Hydroxide.

Reaction Temperature (°C)	Reaction Time (Hour)	mL Sludge Plus mL 10 N NaOH	Recovered Soluble Aluminum (mg/L)	Aluminum Recovery (%)
18	0	10000 + 0	0.04	0
18	0.25	10000 + 100	72	1.2
18	0.50	10000 + 100	138	2.34
18	1	10000 + 100	164	2.78
19	24	10000 + 100	291	4.93

Base = 10-N NaOH
Alkaline-mixing reactor contact time = 1 h
Alkaline-mixing reactor mixing rate = 30 rpm
Volumetric ratio of 10-N NaOH vs. thickened sludge = 700/1000
Alkaline reactor temperature = 20°C
Alkaline reactor effluent treatment = Centrifugation

Reclaimed Soda Alum Quality
Soluble aluminum = 4657 mg/L
Soluble iron = 7 mg/L
THM = ND (nondetected)
TOC = 2652 mg/L
COD = 800 mg/L
Total coliforms = ND
TSS = 2132 mg/L
VSS = 676 mg/L
FSS = 1456 mg/L
Arsenic = ND
Barium = ND
Cadmium = 0.01 mg/L
Chromium = ND
Copper = 1.15 mg/L
Lead = 0.5 mg/L
Manganese = 0.56 mg/L
Mercury = ND
Nickel = 1.31 mg/L
Platinum = ND
Potassium = 14 mg/L
Selenium = ND

Sodium = 4273 mg/L
Titanium = ND
Zinc = ND
pH = 13 units

2.4. *Recycle of Filter Alum and Soda Alum for Water Purification*

2.4.1. *Water Treatment Testing Procedures*

The potable flotation–filtration (sandfloat) water plant in Lenox, Massachusetts, the USA has been continuously serving the Town of Lenox since July 1982, and the reclaimed filter alum and soda alum were not allowed to be fed to the Lenox Water Treatment Plant at the time of this investigation.

Accordingly, the Lenox Institute staff applied the reclaimed alums to a small sandfloat pilot plant at operational conditions (raw water source and quality, DAF clarification rate in gpm/ft^2, chemical dosages, recycle flow rate, and filtration rate in gpm/ft^2) identical to the full-scale Lenox flotation–filtration plant. For the purpose of parallel chemical comparison, the virgin filter alum and virgin soda alum were also dosed to the same Lenox raw water for separate treatment by the same pilot plant unit. The construction of the pilot plant was identical to that of the full-scale Lenox flotation–filtration plant. For each pilot plant testing, water samples were taken 1 h after the system reached its steady state. Experimental results are documented in Tables 7 and 8.

Table 7. Treatment of Normal Lenox Raw Water by Flotation–Filtration Process (Sandfloat) with Recovered Filter Alum and Soda Alum.

Experimental Conditions and Results	Test E (Control)	Test F	Test G
Coagulants (as Al_2O_3)			
Virgin filter alum, mg/L	4.6	0	0
Virgin soda alum, mg/L	1.5	0	0
Recovered filter alum, mg/L	0	5.0	2.8
Recovered soda alum, mg/L	0	8.3	4.7
Flotation–filtration influent quality			
pH, unit	6.7	6.7	6.7
Color, unit	25	25	25
Turbidity, NTU	2.1	2.1	2.1
Aluminum, mg/L Al	0.06	0.06	0.06
Iron, mg/L Fe	0.02	0.02	0.02
Phosphate, mg/L PO_4	0	0	0

(*Continued*)

Table 7. (*Continued*)

Experimental Conditions and Results	Test E (Control)	Test F	Test G
Flotation–filtration effluent quality			
pH, unit	6.6	6.9	6.5
Color, unit	0	0	0
Turbidity, NTU	0.30	0.36	0.27
Aluminum, mg/L Al	0.05	0.10	0.05
Iron, mg/L Fe	0.04	0	0.05
Phosphate, mg/L PO_4	0	0	0

Table 8. Treatment of Lenox Raw Water Containing Phosphate by Sandfloat Process with Recovered Filter Alum and Soda Alum.

Experimental Conditions and Results	Test H (Control)	Test I	Test J
Coagulants (as Al_2O_3)			
Virgin filter alum, mg/L	4.6	0	0
Virgin soda alum, mg/L	1.5	0	0
Recovered filter alum, mg/L	0	5.0	2.8
Recovered soda alum, mg/L	0	8.3	4.7
Flotation–filtration influent-quality			
pH, unit	6.8	6.8	6.8
Color, unit	25	25	25
Turbidity, NTU	2.2	2.2	2.1
Aluminum, mg/L Al	0.06	0.06	0.06
Iron, mg/L Fe	0.02	0.02	0.02
Phosphate, mg/L PO_4	2.02	2.02	2.02
Flotation–filtration effluent quality			
pH, unit	6.6	6.9	6.5
Color, unit	0	0	0
Turbidity, NTU	0.29	0.35	0.27
Aluminum, mg/L Al	0.05	0.09	0.05
Iron, mg/L Fe	0.04	0	0.05
Phosphate, mg/L PO_4	0.03	0.02	0.03

2.4.2. *Experimental Results of Flotation–Filtration Pilot Plant*

Flotation–filtration (sandfloat) Tests E, F, and G (Table 7) were conducted to treat the Lenox Upper Root Reservoir raw water (pH = 6.7; turbidity = 2.1 NTU; color = 25 units; and phosphate = 0 mg/L) using virgin alums and recovered alums. Virgin filter alum, also known as aluminum sulfate or alum, was supplied by Holland Chemicals, Adams, Massachusetts. Virgin soda alum, also known as sodium aluminate, was supplied by Astro Chemicals, Inc., Springfield, Massachusetts.

The results of Test E indicate that the pilot flotation–filtration unit performed as good as the full-scale Lenox flotation–filtration plant when 4.6 mg/L virgin filter alum and 1.5 mg/L soda alum (both as $A1_2O_3$) were dosed. The pilot plant effluent quality was excellent: pH = 6.6 units; color = 0 unit; turbidity = 0.30 NTU; residual Al = 0.05 mg/L; and residual Fe = 0.04 mg/L.

Considering Test E was the control test, the pilot flotation–filtration plant was operated at the same physical and mechanical conditions, but fed with recovered filter alum and recovered soda alum, instead. Experimental results are presented in Table 7 under Tests F and G. In both tests, it appears that the two recovered alums were as effective as the virgin alums for water purification.

To determine whether or not the reclaimed alums can be used for phosphate removal, the raw Lenox Upper Root Reservoir water was spiked with 3.7 mg/L of potassium phosphate K_2HPO_4 (or 2.02 mg/L as PO_4) before the raw water was treated by the pilot flotation–filtration plant. Experimental results are documented in Table 8.

Again a control test (Test H) was conducted using virgin alums (4.6 mg/L filter alum and 1.5 mg/L soda alum; both as Al_2O_3). Two flotation–filtration pilot plant tests (Tests I and J) were conducted using recovered filter alum and soda alum. It was very encouraging to note that both the virgin alums and the recovered alums performed equally well for removal of turbidity, color, and phosphate.

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