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Soil Quality

FOR CROP PRODUCTION

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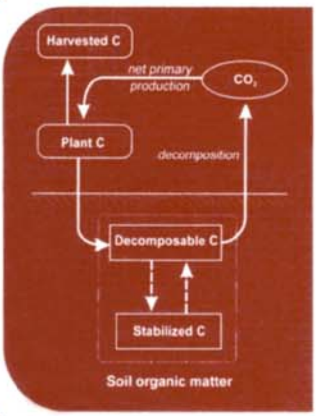

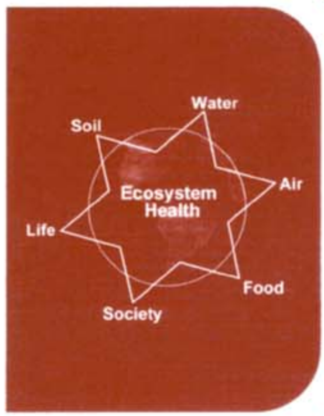

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Developments in Soil Science 25

SOIL QUALITY FOR CROP PRODUCTION AND ECOSYSTEM HEALTH

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"Some seed fell on rocky ground where it found little soil and at once sprang up, because there was no depth of soil; and when the sun came up it was scorched and, not having any roots, it withered away....And some seeds fell into rich soil, grew tall and strong, and produced a good crop..."

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PREFACE

Soils are the interface between aquatic, atmospheric, and terrestrial ecosystems. A better understanding of the linkages between these systems and the role that soils play in those linkages has led to a new approach in assessing soils. Soil evaluation now considers environmental implications as well as economic productivity, seeking to be more holistic in its approach. Thus, soil quality research draws from a wide range of disciplines, blending the approaches of biologists, physicists, chemists, ecologists, economists and agronomists, among others.

The terms "quality" and "health" are somewhat abstract and subjective, and scientists trained to be precise and objective may be uncomfortable using them to frame their work. Nevertheless, these terms capture the broader view that soil research is now striving for. If soil quality is an elusive concept, soil itself defies straightforward definition. Soil is a complex body that exists as many types, each with diverse properties that may vary widely across time and space as a function of many factors. This complexity makes the evaluation of soil quality much more challenging than that of water or air quality. Soil scientists attempt to reduce this complexity by characterizing the properties that can explain more fully the processes taking place in soils. However, because of their wide variability, soil systems can never be fully explained without exhausting every detail in a reductionist way. Such an approach fails to provide the generalizations needed to diagnose problems of degradation or the benefits occurring under different management.

In developing this book we wanted to give as wide a perspective of soil quality as possible. To achieve this, we invited scientists from a range of disciplines to write about what they thought were important factors in assessing soil quality. What is surprising and striking is that common themes and concepts of soil quality should emerge from scientists in different parts of the world working in different disciplines on different soil systems. This encourages us to hope that this holistic approach to soil quality research is bearing fruit and leading in the right direction.

In the first chapter the central concepts of soil quality are introduced. Chapters 2 through 7 focus on specific attributes of soil quality, ecological and pedological views of soil quality, and how soil quality attributes are affected by the process of soil redistribution. Methods to evaluate soil quality, including theoretical discussions of standardization, quality control, pedotransfer functions, and statistics, along with an examination of temporal changes in soil organic matter, are given in Chapters 8 through 12. Practical applications of soil quality assessment, including socio-economic considerations, assessment and predictive tools, and case studies from Australia, Canada, Denmark, and England are described in Chapters 13 through 18. A short glossary provides definitions for terms used throughout the book.

E.G. Gregorich
M.R. Carter

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FOREWORD

The concern of humankind to understand and to maintain soil quality has persisted for thousands of years together with the desire to exploit the soil for the production of food and fiber. At some times, exploitation, with relatively low regard for maintenance of soil quality, has dominated. Volumes have been written about losses of topsoil due to clear cutting of forests on steep slopes, cultivation of large blocks of sloping land, and overgrazing. Desertification is a matter of current concern as growing populations exploit sensitive land for food and fuel. Salinization continues to spread, aggravated by improper irrigation practices. For at least a century, prophets of doom have warned of widespread famine and pestilence as the exponentially increasing human population overwhelms the finite land resource. Some claim that ancient civilizations, such as those in the Tigris–Euphrates basin collapsed largely because of wastage of the land. Others, displaying a level of faith in technology and the corporatist system that would be the envy of any world religion, continue to maintain that technological advances driven by the profit motive will resolve problems of famine, pollution, and disease, and replace the natural environment with improved synthetic environments. As the 21st century dawns, the belief in "technological fixes" may be waning as people become increasingly aware of the need to understand and to live more in harmony with nature.

In 1990, a group of scientists associated with the National Soil Conservation Program of Canada wrestled with the problems of defining soil quality and of methods for measuring its key attributes. For the previous decade or two, related work in Canada had been focused on soil degradation: erosion by wind and water, acidification, salinization, compaction. Half a century before, the emphasis was on stopping wind erosion in the Interior Plains and water erosion in eastern Canada. The idea of focusing on the positive aspect of the soil condition, the quality of the soil, was new to us. We searched the international literature and found that, although aspects of soil quality had been considered for thousands of years, and publications on soil quality were increasingly numerous, no up-to-date synthesis of soil quality research was available. The suggestion was made to Elsevier that such a book be prepared and an agreement was reached. The result of that agreement is this volume that focuses on soil quality as it relates to sustaining crop production and the quality of the environment.

Much of the earlier work on soil quality concentrated on the soil's capacity for crop production. The effects of measures aimed at improving soil quality on surface water quality, groundwater quality, crop quality, siltation of reservoirs, wildlife, and other environmental factors were either not considered or were accorded secondary importance. Thus, for example, rates of fertilizer application were assessed on the basis of crop response and cost with little concern, until a few decades ago, for pollution of groundwater and surface water. Awareness of the environmental aspects of soil quality has been increasing in recent years, and an attempt is made in this volume to give as much consideration to those aspects as to soil quality for crop production.

In the years between the conception and birth of this volume, hundreds of papers and several books on soil quality have been published. The broad scope, international input, and diligence of the editors, however, will make "Soil Quality for Crop Production and Ecosystem Health" a valuable reference for many years. The book should help to convince readers of the importance and fragility of the soil as a component of ecosystems, and of the need to ensure that agro-ecosystems are designed not only to ensure sustained production of food and fiber, but also to accomplish this without degrading adjacent ecosystems.

An improved understanding of the attributes of soil quality and of ways of maintaining or improving soil and associated environmental quality should encourage good husbandry of the soil. Perhaps even more important, this volume should contribute to the growing consciousness of the interrelatedness of all components of Earth, including humankind. Such an awareness of Earth as a system in delicate balance, a system that *Homo sapiens* can destroy, is a first step in the urgent task of changing our goals and our lifestyles so as to be in harmony with Earth and all of its life forms. It is hoped that this book will contribute to awareness and action.

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Chapter 1

CONCEPTS OF SOIL QUALITY AND THEIR SIGNIFICANCE

M.R. CARTER, E.G. GREGORICH, D.W. ANDERSON, J.W. DORAN, H.H. JANZEN, and F.J. PIERCE

I. INTRODUCTION

From the beginning of agriculture, the need to both characterize and assign quality to soil has been self evident. Columella responded to Roman critics of agriculture ("leading men of our state condemning now the unfruitfulness of the soil") with a detailed description and ranking of classes of soils for each of the different kinds of terrain, as well as advice on how to manage them. He pointed out the virtues of good soil husbandry, such as carefully considering the soil moisture status when cultivating, ploughing down residues, and adding manure and incorporating it in order for the soil "to grow fat".

Early scientific endeavours recognized the importance of categorizing soil type and soil variables or properties in regard to land or soil use, especially for agricultural purposes. Soil type and soil properties were usually found to be interrelated and, in combination, could be used to define the *state* of a soil system (Jenny, 1980). Draft requirements for tillage and the cost of inputs (e.g., drainage) to change a soil condition were also factors that determined the quality of a soil for crop growth (Warkentin, 1995). These concerns were further developed by initiating classifications for both land capability and the regional suitability of soils for various uses. These endeavours usually utilized soil properties, landscape factors, and climate to assess the quality of a soil for a specific use.

In more recent years, due to concerns with soil degradation and the need for sustainable soil management in agroecosystems, there has been a renewed scientific attention to soil variables. Coupled with this is the idea of soil use which has emphasized the *value* of soil and soil properties for a specific *function*. Generally, modern concerns with soil quality evolve around the various functions that soils perform in ecosystems. Soil is recognized as a critical component of the earth's biosphere. This ecological approach to soil recognizes soil–human interactions, and the relationship of land managers to soil (Richter, 1987). Thus, soil quality becomes inseparable from the idea of system sustainability, and is considered a key indicator of ecosystem sustainability. The emphasis for soil quality shifts away from suitability for use to whether soil functions are operating at some optimum capacity or level within an ecosystem.

Placing a value upon soil in regard to a specific function, purpose or use leads to the concept of *soil quality*. However, in contrast to water and air, for which the function can be directly related to human and animal consumption, the function placed upon soil is often diverse and usually not directly linked or involved with human health. Thus, the concept of quality is relative to a specific soil function or use. Although soil may have a wide array of possible functions, present day issues have identified three functions of special importance and significance (Doran and Parkin, 1994). Soil functions as a medium for plant and biological production; as a buffer or filter to attenuate or mitigate various environmental contaminants and pathogens; and as a promoter of plant, animal, and, indirectly human health. Table 1.1 provides a more detailed list of commonly accepted soil functions.

Soil quality can only be assessed by measuring properties and therefore involves both an observer and an interpreter. The range of observers, from individuals to interest groups to society as a whole, and the concomitant range in their value systems, ensures diverse views on soil function and consequently on measures of soil quality. For example, Leopold (1949) noted what he termed the A/B cleavage in dealing with soil conservation attitudes. Group A people regard the function of land as production of a marketable commodity, whereas Group B individuals view land in a much broader sense, including not only the biota that it contains, but the functioning of land within the life support system. The latter view recognizes the intrinsic value of soil due to its irreplaceability and uniqueness, and the idea of a relationship between people and soil (Warkentin, 1995).

Functions of soil, and thus soil quality, can be assessed at the field, farm, ecosystem, pedosphere, and global scale. It is recognized, however, that management of soil becomes increasingly difficult at larger scales. Soil, and consequently soil quality, cannot be managed at the global scale. Many aspects of soil quality can be addressed, however, in a practical way at the lower scale. Thus, this introductory chapter will mainly concentrate on soil quality at the lower scale, as it relates to sustaining crop production and improving land management and the quality of the agroecosystem environment.

TABLE 1.1

Examples of functions ascribed to soil that are used to assess quality

Soil Science Society of America (1995)
- Sustaining biological activity, diversity, and productivity
- Regulating and partitioning water and solute flow
- Filtering, buffering, degrading, immobilizing, and detoxifying organic and inorganic materials
- Storing and cycling nutrients and other elements within the earth's biosphere

Larson and Pierce (1994)
- Medium for plant growth and productivity
- Partitioning and regulating of water flow in the environment
- Environmental buffer

Blum and Santelises (1994)
- Biomass production
- Reactor (filters, buffers, transforms matter)
- Biological habitat and genetic reserve

Warkentin (1995)
- Recycling organic materials to release nutrients and energy
- Partitioning rainfall at soil surface
- Maintaining stabile structure to resist water and wind erosion
- Buffering against rapid changes in temperature, moisture, and chemical elements
- Storing and gradually releasing nutrients and water
- Partitioning energy at the soil surface

II. DEFINING SOIL QUALITY

Early concepts of soil quality dealt mainly with various soil properties that contribute to soil productivity, with little consideration of a definition for soil quality itself. However, mere analysis of soil properties alone, no matter how comprehensive or sophisticated, cannot provide a measure of soil quality unless the properties evaluated are calibrated or related against the designated role or function of the soil. Thus, implicit in any definition of soil quality is an understanding of the stated function of the soil, or what the soil does.

In an agricultural context, soil quality is usually defined in terms of soil productivity (e.g., a good quality soil produces abundant, high quality crops), and specifically in regard to a soil's capacity to sustain and nurture plant growth. Thus, from the perspective of agricultural crop production, soil quality can be defined as "*the soil's capacity or fitness to support crop growth without resulting in soil degradation or otherwise harming the environment*" (Gregorich and Acton, 1995).

Although the basic idea behind soil quality is *fitness of a soil for a specific use*, there is an ongoing attempt to more fully define soil quality. Based mainly on a definition of soil fertility introduced by Leopold (1949), Anderson and Gregorich (1984) proposed that soil quality be defined as "*the sustained capability of a soil to accept, store and recycle water, nutrients and energy*." However over the last decade, there has been a definite shift in the way agriculture is viewed. No longer is agriculture viewed as a closed operation, but rather as part of a much broader ecological system, that interacts with, and affects, other various parts of the system. This development is expressed in the expanded concept of soil quality evident in the work of Larson and Pierce (1991). They define soil quality "*as the capacity of a soil to function within its ecosystem boundaries and interact positively with the environment external to that ecosystem.*" This definition also recognizes that soil serves other functions both within and beyond agricultural ecosystems. A more detailed definition has been developed by the Soil Science Society of America (1995) as follows: "*Soil quality is the capacity of a specific kind of soil to function, within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation*". This definition is similar to that of Doran et al. (1996) in which soil quality is the "*capacity of a soil to function, within ecosystem and land-use boundaries, to sustain biological productivity, maintain environmental quality, and promote plant, animal and human health*".

These definitions imply that soil quality has two parts: an intrinsic part covering a soil's *inherent* capacity for crop growth, and a *dynamic* part influenced by the soil user or manager. The latter underlines the lessons of history that good quality soils can be degraded by poor management. Generally, dynamic soil quality changes in response to soil use and management (Larson and Pierce, 1994). The distinction between inherent and dynamic soil quality can also be characterized by the genetic (or static) pedological processes versus the kinetic (or dynamic) processes in soil as proposed by Richter (1987). Koolen (1987) and Carter (1990) also make a distinction between state properties and behavioral properties in soil, which correspond to the concepts of inherent and dynamic as described above.

A. *Inherent soil quality*

The quality of any soil depends in part on the soil's natural or inherent composition, which is a function of geological materials and soil state factors or variables (e.g., parent material and topography). Attributes of inherent soil quality, such as mineralogy and particle size distribution, are mainly viewed as almost static and usually show little change over time. It is generally recognized that some soils have poor natural quality and are not fit or suitable for a specific purpose (e.g., crop production). In some cases, due to adverse management and/or climatic effects (e.g. soil erosion, desertification), human activities such as land use and farming practices can result in the deterioration of a soil that originally possessed good inherent quality (Table 1.2).

Characterization of inherent soil quality for crop production also involves consideration of *extrinsic factors*, those factors apart from soil that influence crop yield (Janzen et al., 1992). These factors include such things as climatic (i.e., precipitation, evaporative demand, and air temperature), topographic, and hydrologic parameters. The latter two factors are often viewed as attributes of landscape quality. Landscape quality plays an important part in land evaluation for a specific use. For example, some land quality attributes (e.g., slope, texture) are used to

TABLE 1.2

Processes associated with land use and management practices that reduce soil quality

Process	Effect on soil attributes/quality	Possible effect on environment
Erosion	Topsoil removed, nutrients lost; capacity to regulate water and energy flow in soil reduced	Deposition of soil material and pesticides in streams and rivers
Loss of organic matter	Soil fertility and structure reduced; capacity to regulate energy flow in soil reduced	Increased soil erosion and degradation, and enhanced greenhouse effect from released CO_2
Loss of structure	Soil porosity and stability reduced; capacity to store and transmit water reduced	Increased runoff and soil water erosion
Salinization	Excess soluble salts and nutrient imbalance; adverse medium for crop growth	Increased bare soil and soil wind erosion
Chemical contamination	Presence of toxins; capacity to act as an environmental buffer exceeded	Movement of chemical via runoff and/or leaching

estimate land quality indicators, such as trafficability, erosion hazard, and drought hazard (Pierce and Larson, 1993).

Generally, inherent soil quality for crop production cannot be evaluated independently of extrinsic factors. For example, a high clay content may be favoured in a semi-arid region, where soil moisture retention is an advantage, but may be undesirable in humid conditions in which poor internal drainage may limit yields. In similar fashion, a certain soil bulk density can be optimal under a semi-arid moisture regime, but deleterious under a humid moisture regime due to changes in relative saturation and subsequent poor soil aeration (Carter, 1990). Because of these considerations, there is no universally applicable set of inherent soil quality criteria and optimum values.

Inherent soil quality can be assessed by using national land resource or soil survey inventories (MacDonald et al., 1995). Huddleston (1984) indicated that the primary reason for initiating soil survey in U.S.A. was for the evaluation of soil productivity, which involved a blend of qualitative and quantitative rating models. Such databases, the fruit of much long-term data collection, can be analyzed in a computerized geographic information system (GIS) to develop broad regional assessments of inherent soil quality and landscape quality (Petersen et al., 1995). Many present databases cover climatic information (e.g., growing degree days), soil texture and depth. Land resource inventories also often include data on soil porosity, nutrient retention, and both physical and chemical rooting conditions (e.g., physical and chemical barriers to root growth).

B. Dynamic soil quality

Dynamic soil quality encompasses those soil properties that can change in response to human use and management. In general, a management system would be viewed as sustainable only if it maintains or improves soil quality (Larson and Pierce, 1994). As implied in the terminology, attributes of dynamic soil quality are subject to change over relatively short time periods. For example, total organic matter may change over a period of years to decades, whereas pH and labile organic matter fractions may change over a period of months to years. In comparison, microbial biomass and populations, soil respiration, nutrient mineralization rates, and macroporosity can change over a period of hours to days. Thus, maintenance and/or improvement of dynamic soil quality deals primarily with those attributes or indicators that are most subject to change (e.g., loss or depletion) and are strongly influenced by agronomic practices. Table 1.3 provides examples of various soil processes on the basis of time scale.

Present approaches to quantify soil quality are concerned with either directly characterizing different attributes of quality (i.e., soil properties) or identifying specific indicators that can represent the attribute in question (Gregorich et al., 1994).

C. *Soil versus land quality*

As mentioned above, evaluating inherent soil quality involves the consideration of extrinsic factors, including landscape quality. In the past there has been some confusion in separating definitions of soil quality versus land quality (Pettapiece and Acton, 1995). It is now recognized that soils are part of a larger environmental system, and that *land* is a term that better reflects the natural integration of soil, water, climate, landscape, and vegetation characteristics (FAO, 1976; Hamblin, 1995; Pettapiece and Acton, 1995). Thus, soil quality is one component of land quality.

The distinction between soil and land quality can also be illustrated by the relatively recent pedological focus on soil as a spatial three-dimensional entity (Lavkulich, 1995). In this comparison, pedology has developed from site analysis to

TABLE 1.3

Examples of various processes in soil according to time scale

Long term (10^2 to 10^3 yr)	Medium term (1 to 10^2 yr)	Short term (seconds to 1 yr)
Humus decomposition	Clay formation	Evaporation
Podzolization	Clay destruction	Carbonate leaching
Gleying	Clay transformation	Heat transport
Laterization	Pseudogleying	Gas diffusion
Solodization	Erosion	Ion exchange
	Salinization	Mineralization
		Immobilization
		Compaction
		Loosening
		Desalinization

two-dimensional soil profile analysis, to three-dimensional pedon analysis, with more recent emphasis on three-dimensional polypedon or landscape units. These developments are reflected in the cartographic concept of a catena. Spatial aspects are also being emphasized by the interest in GIS and spatial statistics.

Land quality, then, is a broader concept than soil quality as it describes the state of the combined or integrated entity of soil, water, and vegetation (FAO, 1976; Hamblin, 1995). Similar to soil quality, land quality can be characterized by attributes and indicators. The development of land quality indicators, the key entities that can be analyzed and monitored to assess land quality, is currently the subject of much research.

D. Concept of soil health

It has long been recognized that there is an indirect relationship between soil health and the health of animals and humans, via the quality of crops (Warkentin, 1995). Here, soil health is mainly concerned with the balance and availability of plant nutrients, and freedom from plant diseases and pests. More recently, the above approach to soil quality has led to the idea of characterizing a soil's health (Larson and Pierce, 1991; Doran and Parkin, 1994; Doran et al., 1996). In this development, the soil is seen as a living system, and the functions placed upon soil can be viewed in terms of "goals", "potentials", "normal states" or "optimum states", and their condition (i.e., health) assessed and compared against some standard. In addition, the ability of a soil to handle stress and to recover equilibrium (i.e., resilience) after some form of perturbation can also be viewed in terms of "health". Thus, in this approach soil health very closely parallels soil quality.

The concept of "health" can be considered at many levels of biological organization, from the individual (i.e., organism) to a community of individuals (i.e., system) (Rapport, 1992; Ferguson, 1994; Callicott, 1995; Park and Cousins, 1995). In the former, health is a composite picture of the condition of an organism's (e.g., human body) various parts and functions, and it is assessed by characterizing many different factors. In the same way, using the soil-as-an-organism analogy, soil health is a composite picture of the state of the soil's many physical, chemical, and biological properties, it's shape and morphology, and the processes that interact to determine this quality. For the "system" or "community-of-individuals" analogy, health implies a viable and self-sustaining condition, a system that is producing the "goals" and maintaining the "values" placed upon it by society (Rapport, 1992). The concept can be applied at the agroecosystem (Gallopín, 1995) and ecosystem (Callicott, 1995) levels. Generally, the soil-as-a-community analogy is of greater utility for describing soil health than the soil-as-an-organism analogy (Doran et al., 1996).

Although the concepts of soil health and quality may seem synonymous, there are situations where they differ. For example, as noted by Sparling (1997), a soil with poor quality for a specific purpose (i.e., crop production), such as a sand, may nevertheless be healthy. That is, the sand, in its natural setting and within the bounds of its inherent composition and properties, can function at some optimum state as a vital living system. Generally, the basic idea behind soil quality as "fitness of a soil for a specific use", especially in the context of inherent soil quality as defined earlier, will undoubtedly result in otherwise healthy soils being designated as "poor quality".

Overall, just as there is no single measure of organism or community health, there is no single measurement that can be used to quantify soil health. It must be inferred or estimated by using the same framework (i.e., function, process, attributes, and indicators) as described for soil quality in the next section. Doran et al. (1996), using similar terms as soil quality, defined soil health as "*the continued capacity of a soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain biological productivity, maintain the quality of air and water environments, and promote plant, animal, and human health*".

E. Soil quality and land sustainability

Soil quality is considered to be a key element of sustainable agriculture (Warkentin, 1995). The latter can be defined as "*agricultural and agri-food systems that are economically viable, and meet society's need for safe and nutritious food, while conserving or enhancing natural resources and the quality of the environment for future generations*" (Globescan, 1996). Overall, sustainability refers to productivity, economic, social, and environmental aspects of land use systems (Smyth and Dumanski, 1995). Generally, the main areas involved with agricultural sustainability are as follows: maintaining or improving farm productivity; avoiding or minimizing adverse impacts on natural resources and associated ecosystems; maximizing the net social benefit derived from agriculture; and promoting flexibility of farming systems to manage the risks of climate and markets. Thus, although sustainability issues are much broader than soil quality, the strong emphasis on maintaining the natural resource base ensures that maintaining good soil quality is an integral part of sustainable agriculture (Miller and Wali, 1995; Royal Commission on Environmental Pollution, 1996).

In contrast to natural ecosystems, in which equilibriums are established that reduce export of nutrients from the system, agroecosystems are characterized by management inputs that increase nutrient recycling and decrease natural soil buffering (Warkentin, 1995). Addiscott (1995) suggested that the principle of minimum entropy production characterizes natural ecosystems, whereas agroecosystems tend towards degradation of ordered complex structures, resulting in production of environmentally undesirable small molecules (e.g., N_2O). Thus, to achieve sustainability, agricultural systems should be managed to produce a steady state; to maintain the capacity for self organization (i.e., maintain biological potential); and to inhibit dissipative processes that produce excess small molecules, because these molecules often cause environmental or human health problems.

As indicated earlier, the concept of "quality" implies purpose, use and value. In natural ecosystems this concept does not readily apply, as "better" and "worse" environments do not exist (Hamblin, 1995), although most natural ecosystems are influenced in some degree by anthropogenic processes (e.g., change in climate and/or biodiversity). Generally, the means to assess agricultural sustainability is by identifying indicators that reflect an attribute or set of attributes of sustainability. Such indicators are considered tools to both *warn* of deleterious environmental changes and to *compare* farming practices (Standing Committee on Agriculture and Resource Management, 1993; Environmental Indicator Working Group, 1994). Key indicators for agricultural sustainability, some of which are related or linked to soil function, are selected to describe the following: changes in long-term real net farm income; changes in land and water quality to sustain production at levels set by climate and land capability; changes in the level of managerial skills of land managers in finance, farming practices, and environmental stewardship; and changes in food quality and off-site landscape hydrology and native ecosystems caused by agricultural practice.

III. EVALUATING SOIL QUALITY

A useful framework to evaluate soil quality is based on the following sequence: functions, processes, attributes or properties, attribute indicators, and methodology. Soil quality is evaluated on the basis of the function in question. Functions deal with "what the soil does", or "what the soil is asked to do". Each function can be characterized by specific soil processes that support the function being imposed upon the soil. Soil quality attributes can be defined as measurable soil properties that influence the capacity of the soil to perform a specific function (Acton and Padbury, 1993). Generally, attributes describe a critical soil property involved with the process or processes underlying a function. The attribute, or soil property, is most useful when it reflects or measures *change* in the process. In many cases the specific property may be difficult to measure directly, so an indicator (an associative property; i.e., a surrogate or proxy) or pedotransfer function (a related property; Bouma, 1989) can be used to serve as an indirect, practical measure of the attribute. Indicators can represent a single attribute or a set of attributes. It is generally acknowledged that indicators should be easily measured and verifiable, have some sensitivity to variations in soil management (but not be overly sensitive), and have a relatively low sampling error. Indicators that have a relatively long record of sampling or are found in historical records are of particular use. The choice of indicator would be based on the provision of available methodology, including ease of duplication and facility for accuracy and speed. Table 1.4 illustrates the above approach, and the following subsections address specific aspects of the framework to evaluate soil quality.

A. Functions of soil for crop production

With respect to crop production, the function of soil is to nurture and sustain plant growth. This function is related to the efficiency with which soil provides essential nutrients, substrates, and environment to support the conversion of CO_2 to organic molecules using energy from sunlight (via photosynthesis). The function of soil for crop production can be subdivided into several components as follows:

TABLE 1.4

Example of a framework (given in part) for evaluating soil quality: characterizing some of the capacities of a soil to perform a specific function (i.e., provide a medium for plant growth)

Process	Attribute or property	Indicator for attribute	Possible method for measuring attribute
Capacity to accept, hold, and release water	Infiltration	Infiltration rate, sorptivity	Tension permeameter
	Water-holding capacity	Desorption curves	Tension table, pressure plate
	Permeability	Hydraulic conductivity	Guelph permeameter
Capacity to accept, hold, and release energy	Organic matter	Organic carbon	Dry combustion
	Labile organic matter	Microbial biomass	Chloroform fumigation
		Carbohydrates	Acid hydrolysis
		Macroorganic matter	Dispersion/sieving
	Particle size	Clay	Hydrometer/pipette

provide a medium of plant growth; regulate and partition water, gas and energy flow, and serve as a buffer or filter system. Evaluating these function components, to assess a soil for its quality to produce crops involves considering the soil's chemical, physical, and biological properties. Table 1.5 lists the functions of a soil related to plant growth, and some of their characteristics.

TABLE 1.5

Characterizing the main function components of a soil in regard to crop production

Function component	Function characteristics/processes
Medium of plant growth	Suitable medium for seed germination and root growth
	Absence of adverse chemical conditions (acidity, salinity, sodicity)
	Supply balance of nutrients
	Suitable medium for microbes (nutrient cycling, decomposition)
	Promote root growth and development
Regulate water	Receive, store, and release moisture for plant use
	Adequate water retention to buffer and reduce effects of drought
	Adequate infiltration and storage capacity to reduce runoff
Regulate gases	Accept, hold, and release gases
	Adequate air movement and exchange with atmosphere
Regulate energy	Store release (recycle) energy rich organic matter
Buffer or filter	Accept, hold, and release nutrients
	Sequester energy compounds and/or biotoxic elements
	Detoxify substances harmful to plants

B. Attributes of soil quality

Numerous soil properties can serve as attributes of soil quality for crop production and agricultural sustainability. The challenge is to identify soil properties that reflect the capacity of the soil to generate and sustain plant growth, as the link between some soil properties and crop response may have little empirical basis. For example, soil organic matter may explain a significant proportion of the variability in crop yield but is limited as a sole indicator of overall soil productivity (Janzen et al., 1992). This is because soil productivity is a result of multiple variables of which organic matter is but one. In a similar fashion, many soil physical properties act only indirectly on crop growth. Soil bulk density, for example, is often poorly related to plant yield, but does influence several other soil properties (e.g., strength, permeability, water retention) that can individually or collectively impact directly on crop productivity (Koolen, 1987; Carter, 1990). This characteristic, common to many important soil properties, should be considered in the quest to identify potential soil quality attributes.

Various studies have attempted to identify sets of attributes or properties that can characterize a soil process or processes in regard to a specific soil function (Larson and Pierce, 1991; Arshad and Coen, 1992; Gregorich et al., 1994; Karlen et al., 1994; Linden et al., 1994; Turco et al., 1994). A major goal in soil quality studies is to ascertain, where possible, links between properties (or indicators/proxies/surrogates) and a specific function of the soil (e.g., crop productivity). Once a property is identified for a specific soil type or situation, information about *soil quality standards* for a given set of conditions is needed. This involves information on the critical ("threshold") level and range of the attribute (property) associated with significant changes (usually adverse) in the soil function of interest (e.g., optimum crop production). Development of soil quality standards can be a difficult process, especially defining the limits or critical range (Pierce and Larson, 1993). In response to these concerns, the International Organization for Standardization (ISO) is developing various standards for soil quality measurements that address the different phases (e.g., soil sampling, handling, storage, analysis) involved in soil characterization (Hortensius and Welling, 1996; Nortcliff, this volume).

C. Minimum data sets

Identifying key soil attributes that are sensitive to soil functions allows the establishment of *minimum data* sets (MDS) (Larson and Pierce, 1991, 1994). Such data sets are composed of a minimum number of soil properties that will provide a practical assessment of one or several soil processes of importance for a specific soil function. Ideally, the property should be easily measured, and the measurements reproducible and subject to some degree of standardization. In cases where the property of interest is difficult or expensive to measure, an indicator or pedotransfer function may provide an alternative estimate (Table 1.6).

Different soil processes require different MDSs, although they may contain some common attributes or properties. In addition, MDS can be developed to provide a compilation of sub-attributes to further describe a specific attribute, such as sub-attributes or properties to describe the multi-faceted role or function of soil organic

TABLE 1.6

Minimum data set of soil chemical, biological, and physical attributes (properties), with selected indicators and variables of pedotransfer functions, to assess the main functions of soil for plant growth (adapted in part from Larson and Pierce, 1991, 1994)

Soil attribute or property	Indicator	Pedotransfer variables[1]	Methodology
Nutrient availability	Soil extractants	Plant accumulation	Soil test
Adsorbed nutrients	Cation exchange	Clay type + org. C	Displacement
Organic matter	Organic C	Clay + silt	Combustion
Texture	Hand or "feel" method		Particle size analysis
Available water	Water constants	Particle size & org. C	Desorption curve
Structural form	Density, porosity	Particle size & org. C	Bulk density
Structural stability	Aggregate stability	Clay and org. C	Wet sieving
Strength	Penetrability	Density & water	Penetrometer
Rooting depth	Penetrability	Bulk density	Observation of roots
Reaction	pH		Electrode
Soluble salts	Electrical conductivity	Plant growth	Conductivity meter
Sodicity	Sodium adsorption ratio	Soil strength	Saturation extracts

[1]Variable used in pedotransfer function

matter (Gregorich et al., 1994). The MDS can be expanded to assess and accommodate a much broader scheme, such as soil health and ecological concerns (Doran and Parkin, 1994). An example of the former is given in Table 1.7.

D. Assessing change in soil quality

Dynamic soil quality for crop production is concerned with changes in soil quality attributes or properties resulting from land use and management. One of the goals of

TABLE 1.7

Minimum data sets of (sub) attributes of soil organic matter to address different soil processes (after Gregorich et al., 1994)

Process	Minimum data set
Soil structural stability	Total organic C Microbial biomass Carbohydrates
Nutrient storage	Total organic N Microbial biomass N, and mineralizable N Light fraction and macroorganic matter
Biological activity	Microbial biomass Enzymes Mineralizable C and N

sustainable agricultural systems is to maintain soil quality. Thus, evaluating soil quality, in addition to characterizing functions, identifying attributes, and developing MDS also requires strategies to evaluate soil quality change. Larson and Pierce (1994) discuss both the *comparative assessment* and *dynamic assessment* approach to evaluate soil quality change. The former is commonly used and involves a single comparison of one system against another. However, although this approach may provide a measure of change over a specific time span (if the initial data are available), it gives little information on trends in soil quality or rates of quality change over time. In contrast, dynamic assessment compares or evaluates soil quality attributes continuously over time. Larson and Pierce (1994) identify both computer models (which use attributes as variables) and statistical (i.e., temporal pattern of attribute mean and standard deviation) control as a means to assess soil quality change over time. Other approaches are use of archived soil and plant samples from long term experiments, and geostatistical methods.

In many cases, dynamic assessment requires a monitoring system to provide a regular surveillance of soil quality attributes or indicators. Monitoring has been commonplace for air and water but not for soil. Some studies, such as soil erosion, have employed monitoring to estimate rate of soil loss but information on change in soil productivity or other quality measurements are generally not obtained (Pierce, 1996). In Canada a soil quality monitoring system was initiated in 1989 (Wang et al., this volume) and data on crop productivity and changes in soil quality are being collected.

In regard to change in soil quality, standards are needed to assess if the recorded changes are within natural variation or optimum range of the soil attribute in question, or if the changes are related to management practices that may require changes if quality is deteriorating. Since within a minimum data set individual attributes or indicators may show opposite or different changes (e.g., organic matter increasing, but porosity decreasing), the interpretation of such changes and the required management response underlines the importance of 'experience' and 'skill' in the soil manager.

IV. SOIL QUALITY FOR IMPROVED LAND MANAGEMENT

The goal of the land manager is to sustain and improve the quality of the soil resource base. Thus, soil quality is in the hands of the land manager (Pierce and Larson, 1993; Pierce, 1996). Monitoring soil quality does not in itself change the soil condition, but serves only to indicate if changes in management are required. Monitoring is important but the usefulness of the data will only be realized if it is used in management decisions to correct deficiencies or improve the quality of the soil resource.

Land managers need well designed soil and land management systems and quality control procedures to ensure that those processes that are important to soil quality and at the same time responsive to management are operating at an optimum level. Therefore, soil quality control in soil management involves both monitoring and regulation. The latter activity relates to the continued application of management inputs and improvements to ensure that soil quality is not deteriorating (Pierce, 1996). For example, monitoring may indicate that organic matter levels in a specific soil are in decline or too low to resist soil erosion. Application of the regulatory aspect of soil quality control could proceed in two parts as follows: 1) assess if the soil management system is capable of producing or providing adequate organic matter (i.e., inputs of crop residue and/or organic amendments) to prevent a decline in soil organic matter, and 2) assess if management strategies allow the best use or placement of present organic matter inputs. A negative conclusion in the first part would indicate a non-sustainable land management system and emphasize the need for change in the agricultural system; a negative in the second part may call for better management practices, such as use of a cover crop and/or a change in tillage tool components to improve crop residue cover.

In reference to the above concerns, Pierce and Larson (1993) emphasize that sustainable land management should include the following assessment: evaluate land suitability for specific use, identify key soil quality attributes for the specific system and derive a minimum data set, establish soil quality standard limits, identify management inputs that strongly influence soil quality attributes (e.g., residue levels influence soil organic matter), employ soil quality control techniques to monitor the system, and modify management as needed to maintain soil quality control. The recent emergence of site-specific management concepts and technologies (Robert et al., 1995) is an important step forward in soil quality control procedures (Pierce, 1996).

V. SOIL QUALITY AT THE ECOSYSTEM LEVEL

As noted earlier, soil can serve various functions besides its role in crop production and agriculture. At a higher level, soil performs certain functions in ecosystems and at a global scale (e.g., Table 1.1). All of these expanded functions of soil can be evaluated on the basis of "quality" as described in earlier sections. However, it is recognized that "quality" relates to the role given to the soil. Thus a soil of high quality for agriculture may be of suboptimal quality from an ecosystem or global perspective.

Terrestrial ecosystems contain soil, atmosphere, water, vegetation, and animals. As components of ecosystems, soils function to both regulate biotic processes (e.g., supplying plants with mineral nutrients and water) and flux of elements (e.g., turnover and storage of C, N, P, and S). These soil functions also affect other components of the ecosystem (i.e., aquatic, atmospheric, and biological), as well as adjacent ecosystems. Soil alters the chemical composition of precipitation and distributes water through the environment; contributes to the gas, water, and heat balances of the atmosphere; and serves as a reservoir of biodiversity and genetic material. Table 1.8 illustrates some of the wider functions of soil at both the ecosystem and global levels.

The fundamental unit for assessing soil quality at the ecosystem level is usually the soil horizon. The properties of soil horizons (e.g., thickness, organic matter content, pH) are used to characterize the pedon. At higher levels or scales of organization,

TABLE 1.8

Characterizing the function of soil at the ecosystem and global scale

Function	Example of function characteristic
Accumulation and store of biogenic energy	Storage of energy rich organic matter and regulator of C, N, P, S cycle
Maintenance and storage of organic matter	Storage of active, cycling organic biophilic elements
Mitigation of toxic elements	Sequestration of soluble Al in aluminum-organic complexes
Mitigate accumulation of atmospheric CO_2, NO_x, and CH_4	Terrestrial pool of organic carbon and a source and/or sink for CO_2, NO_x and CH_4
Hydrological cycle	Water storage, runoff, infiltration, and leaching
Buffer climatic transitions	Dampen daily and seasonal oscillations of temperature and moisture

pedons can be grouped into catenas, soil types, landscape units, zones, and ecozones. Levels or scales of soil organization below the horizon, such as aggregates and organo–mineral colloids, control the internal processes or physiology of soil. Understanding these processes is fundamental to understanding the higher levels or scales of soil organization and the overall functioning of soil in an ecosystem. For example, regional levels of greenhouse gas emissions cannot be estimated based on soil type alone without knowledge of decomposition and denitrification processes at the microsite level.

Overall, the value of considering soil quality at the ecosystem level is that it provides an integrative approach and examines the function of soil in its natural context. It allows detailed investigation of specific soil properties and processes, but requires that results be interpreted in relation to the whole.

VI. SUMMARY

Although soil quality can be simply defined as a soil's "fitness for use", it is in reality a complex concept and significantly more challenging in its assessment than air or water quality. Soil quality can basically be divided into inherent and dynamic quality. The former is a component of land quality, whereas the latter is strongly influenced by the soil manager. Measurement of soil quality involves placing a value upon soil in relation to its fitness to perform a specific function or purpose. Functions can vary in relation to both use of soil and scale. Once a function has been established, it is possible to identify and characterize soil processes and attributes that describe the function, the indicators that are related to the attribute(s), and methodologies for measuring these. This allows the development of soil quality standards and control techniques, and subsequently the design of sustainable land management systems. Overall, the following conclusions can be given in regard to the concept of soil quality:

- The concept of soil quality is not altogether new, but is undergoing development in response to the idea that soils are part of land or terrestrial ecosystems. Thus, soil quality brings together old and new ideas about soil and land.
- It is important to recognize the difference between inherent and dynamic soil quality, as well as the difference between soil and land quality. Further, although soil quality describes an objective state or condition of the soil, it is also subjective or evaluated partly on the basis of personal and social determinations.
- Ecosystem concepts such as function, processes, attributes, and indicators, provide a useful and robust framework to describe soil quality. This framework is also useful when it is directed towards the intensive manipulation, engineering, and/or management of the soil resource.
- In the context of using soil intensively as a resource, soil quality becomes a technology or an applied science, directed towards problem solving (e.g., better soil management) and can be seen as a key to sustainable land management.
- The basic idea of "fitness for use" in regard to agricultural and/or industrial use of soil, reflected in early attempts to classify "soil suitability" or "land capability", is the basic premise of soil quality. If a soil is not suitable for a specific use then it is not appropriate to assign or describe quality for that specific use or function. In many cases, however, it is not possible to make a perfect match between the soil and its intended use, and quality must be built into the system.
- A large range of attributes, such as chemical, physical, and biological properties, can be used to describe soil quality. Attributes need to be selected for specific soil uses. However, some attributes have a wide utility and can serve a wide range of purposes. Thus, a "minimum data set", composed of a limited number of key attributes, is the usual approach in soil quality investigations, except for some singular situations (e.g., disturbed hydrology) in which a dominant soil response can be characterized using a single attribute.
- A major impediment to the evaluation of soil quality is the lack of standardization, related to both methodology and "critical limits". Soil quality standards are required to ensure that soil sampling, description, and analysis can set the limits for a quality soil and detect adverse changes in soil quality.

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Chapter 2

PHYSICAL ATTRIBUTES OF SOIL QUALITY

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I. INTRODUCTION

Crop production and ecosystem health are affected strongly by soil physical quality, particularly in the root zone. The soil must provide a foundation that is sufficiently strong to provide adequate plant support and stable soil structure, but not so strong as to inhibit root proliferation and faunal activity, such as burrowing. Under these soil mechanical conditions (and assuming climate and crop factors are not limiting), the additional requirements for crop growth are optimal supplies of water, dissolved nutrients, oxygen, and heat. In addition, the soil environment should be such that soil flora and fauna can flourish, leaching of crop nutrients and pesticides out of the root zone are minimized, and agricultural activities (e.g., tillage, planting, harvesting) can proceed in a timely manner without damaging soil structure and compacting the soil. Soil physical conditions that allow and/or promote these responses constitute "good soil physical quality".

Soil physical quality derives mainly from primary and secondary soil particles and the voids or pore spaces within and between those particles. Primary soil particles include rock and mineral fragments and organic matter deriving from both living and dead soil flora and fauna. Secondary soil particles are essentially clusters of primary particles (usually referred to as aggregates, peds, or clods) that also contain mixtures of soil biota, organic material, water, and air. The nature and relative proportions of the primary and secondary soil particles in turn determine the amount, morphology, continuity, and degree of interconnection of the pore spaces within and between the soil particles. The collective nature of the soil particles and the associated pore spaces (often referred to as the soil's "structure") impart to the soil two fundamental physical attributes: capacitance and strength. This chapter focuses on the capacitance and strength attributes of soil, and on how these attributes affect crop production and ecosystem health.

II. SOIL CAPACITANCE AND STRENGTH

A. Soil capacitance

Soil capacitance relates to the ability of the soil to store and transmit liquids, solutes, gases, and heat. It is determined by the amount and nature of the pore spaces within and between the soil particles. It is within the pore spaces that the biological, chemical, and physical processes take place, while the matrix, made up from the particulate material, provides the physical structure and stability.

The storage component of soil capacitance is determined primarily by the total amount of pore space or "porosity" of the soil. Soil porosity (ε) is defined as the fractional volume of void or pore space in the soil (V_v) divided by the total volume of soil (V_t), which includes the volumes of soil solids (V_s), soil water (V_w), and soil air (V_a):

$$\varepsilon = \frac{V_v}{V_t} = \frac{(V_a + V_w)}{(V_s + V_a + V_w)} \tag{1}$$

The magnitude of ε depends on the size distribution of the primary soil particles (i.e., soil texture), as well as on the amount, nature, and size distribution of the secondary soil particles (i.e., soil structure). For example, medium-textured, structureless soils (e.g., compacted sands, silts, loams) tend to have lower porosity than fine-textured, structured soils (e.g., uncompacted loams and clays). Generally speaking, the greater the porosity the greater the soil's ability to store liquids, solutes, gases, and heat.

The transmission component of soil capacitance is determined by the morphology, continuity, and interconnectedness of the pore spaces. The intrinsic ability of the soil to transmit any fluid, such as water and air, is described by the soil permeability, k. The k parameter, which depends only on the physical characteristics of the soil pores and not on the type of fluid flowing, is often represented by:

$$k = Cr^2 \tag{2}$$

where C is a dimensionless proportionality constant and r is the mean soil pore radius. The pore characteristics, other than the radius, that affect fluid transmission are included in C, such as pore size distribution, pore roughness, pore shape, pore tortuosity, and continuity. The greater the k of a soil, the greater ability of the soil to transmit any fluid. Optimum fluid transmission through the soil is usually achieved when a wide range of pore sizes exist, rather than when there is a predominance of a particular pore size, (i.e., a wide or well graded pore size distribution is better than a narrow or poorly graded pore size distribution). For example, soils with a predominance of small pores (e.g., poorly structured clayey soils) have a low permeability and consequently tend to suffer from restricted drainage and poor aeration (water logging), whereas soils with a predominance of large pores (e.g., sandy soils) have a high permeability and thereby tend to suffer from excessively rapid drainage and droughtiness. In contrast, soils with a well graded pore size distribution can both rapidly drain off excess water, thereby minimizing poor aeration, and retain water essential to the crop, thereby preventing droughtiness.

Porosity and permeability allow the soil to be "charged up" with liquids, solutes, gases, and heat, much like an electrical capacitor or battery can be charged up with electrical potential. The amount of "charge" the soil can accept and the rate at which the soil can be "charged" increase with increasing porosity and permeability, respectively. For example, a dry soil that is both highly porous and highly permeable can rapidly accept a large volume of rainfall or irrigation water. A soil that has high porosity but low permeability, on the other hand, can also accept a large volume of water, but only at a low rate of addition (i.e., high rates of water addition only result in water loss through surface runoff). The porosity and permeability also allow the soil to "leak" or "bleed off" its supply of liquids, solutes, gases, and heat in much the same way as a capacitor or battery gradually loses its electrical charge (discharges). The degree and rate of bleed off or discharge is again affected largely by the magnitude of the porosity and permeability, but also by the consumptive demands placed on the soil by plants, soil organisms, and the atmosphere.

B. Soil strength

From an agricultural perspective, soil strength relates to the ability of the soil to: i) resist the loss of structure through tillage-induced pulverization or remoulding, traffic-induced compaction, and rainfall- or irrigation-induced slaking (breakdown) of aggregates; and ii) resist penetration by crop roots and burrowing soil fauna. A soil with good physical quality should be strong enough to maintain its structure and hold plants upright, but also weak enough to allow extensive penetration by crop roots, soil flora, and soil fauna.

The main components of soil strength include shear strength, compressive strength, and tensile strength. In simplistic terms, shear strength is the ability of the soil to resist lateral movement; compressive strength is the ability to resist compaction (reduction in total porosity) due to forces acting inwardly; and tensile strength is the ability to resist expansion (increase in total porosity) due to forces acting in the outward direction. All three aspects of soil strength arise to a greater or lesser extent from the packing and frictional resistance among soil particles, the electrostatic attraction (cohesion) between clay minerals, the presence of organic and mineral binding agents, and the inter-particle adhesion resulting from the presence of water. Generally speaking, soils that display good strength are those that have well-graded (wide) particle size and pore size distributions (e.g., structured loam soils), along with ample organic matter in various stages of decomposition.

C. *Interaction of capacitor and strength attributes*

The capacitor and strength attributes of the soil interact in a complex manner. Improving one attribute may cause another to decline. For example, improving the soil's ability to store water may decrease its ability to store air (Eq. 1). Increasing permeability improves the soil's ability to replenish the root zone with rainfall or irrigation water, but the increased permeability may also cause a more rapid loss of water and dissolved agrochemicals from the root zone because of more rapid internal drainage. Reducing soil strength may initially improve root proliferation in the root zone, but it may also promote loss of soil structure and allow compaction, which will in turn reduce the soil's ability to store and transmit fluids. Establishing good soil physical quality for maximum crop production and ecosystem health therefore involves a careful optimization of all of the main soil physical properties and attributes.

The next four sections of this chapter describe the capacitor and strength attributes of the soil. These sections also present various critical soil water, aeration, strength, and structure parameters that might be used to measure soil physical quality and to monitor the evolution of soil physical quality in response to changes in agricultural production. The last section discusses briefly the need to combine various critical soil parameters to obtain higher-level soil physical quality indicators that reflect the strong and synergistic interactions between soil, agricultural activities, crop growth, and weather.

III. SOIL WATER

An adequate volume and rate of supply of soil water is obviously crucial for maximum crop production and the existence of many essential root-zone organisms. Soil water not only provides the bulk of the water necessary for the function of living cells in and on the soil, but it is also the major vehicle by which plants and soil organisms absorb nutrients and eliminate waste products (e.g., Hausenbuiller, 1978). The presence of soil water in excess of requirements can, however, be detrimental by causing poor aeration and/or excessive leaching of soil nutrients and pesticides out of the root zone (primarily in humid climates), or by causing the accumulation in the root zone of phytotoxic salts (primarily in arid climates). A soil with good physical quality must therefore not only provide sufficient water for crop and soil organism requirements, but also quickly eliminate the potentially harmful excess water. The extent to which a soil can achieve this is determined by its water storage and transmission relationships.

A. Water storage

The volume of water present or "stored" in soil is described by the volumetric water content or retention relationship, θ (ψ):

$$\theta(\psi) = \frac{V_w(\psi)}{V_t}; \qquad \psi \precsim 0 \tag{3}$$

where θ is volumetric water content (volume of water per unit bulk or total volume of soil), ψ is the soil pore water pressure head or matric potential, V_w is the volume of water present, and V_t is the total soil volume which includes the volume of soil solids, water, and air.

As indicated by Equation 3, $\theta(\psi)$ is not a single value, but a monotonic relationship with θ decreasing as ψ decreases (becomes more negative). This occurs because decreasing ψ is equivalent to increasing the tension or suction on the water in the soil, which in turn causes progressively more water to be pulled from the soil. As ψ is decreased, those pores having the largest radius are drained first, and the relationship between ψ and θ can be related to a pore size distribution.

Because of the inherent complexity of the $\theta(\psi)$ relationship, and because it is also difficult to measure in the field, $\theta(\psi)$ is often approximated by the "soil water desorption" or "main drainage" curve determined in the laboratory from soil cores. The desorption curve is obtained by first saturating the soil core, and then measuring the equilibrium θ at successively lower (more negative) values of ψ (Topp et al., 1993). Example desorption curves are given in Figure 2.1, where θ at $\psi = 0$ (i.e., the saturated volumetric water content, θ_s) equals the soil porosity, ε (Eq. 1), because the volume of soil air is zero at complete saturation. In the field, both wetting and drying occur or absorption and desorption occur in rapid succession, adding complexity to the observed $\theta(\psi)$ relationship. Fully saturated conditions seldom occur above the groundwater, and the $\theta(\psi)$ relationship of a wetting soil is somewhat lower than obtained by desorption as in Figure 2.1.

Two important points on the $\theta(\psi)$ curve are the "field capacity" and "permanent wilting point" water contents (Fig. 2.1). Field Capacity (FC) is the volumetric water content (θ) obtained when a soil is initially saturated and then allowed to drain under gravity until a quasi-static equilibrium water content is reached, a process that usually requires one to three days. The pressure head corresponding to FC, ψ_{FC}, depends to some extent on climate and water table depth (Cassel and Nielsen, 1986; Hillel, 1980b; Koorevaar et al., 1983) and is consequently often given as $\psi_{FC} = -5$ kPa for cool regions with high rainfall and shallow water tables (e.g., Great Britain and Northern Europe), as $\psi_{FC} = -10$ kpa for temperate regions with intermediate rainfall and water table depths (e.g., central Canada and U.S.A.), or as $\psi_{FC} = -35$ kPa for dry regions with deep water tables (e.g., semi-arid plains areas of Canada, U.S.A., and Australia).

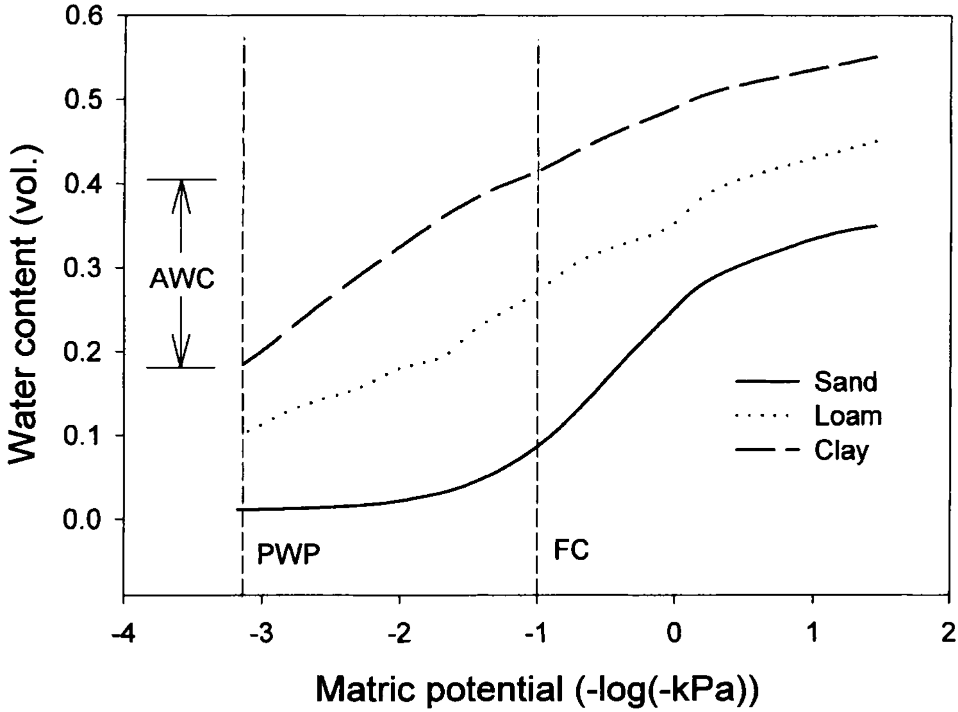

Fig. 2.1. Examples of typical desorption relationships for different soil textures. FC is field capacity, PWP is permanent wilting point and AWC is available water capacity.

Permanent Wilting Point (PWP) is the volumetric water content (θ) below which most crops cannot extract any more water from the soil. When crops reduce θ to the PWP they permanently "wilt" and begin to die unless water is quickly added to the soil through precipitation or irrigation. The precise determination of PWP is complex and depends on the crop as well as the soil type (Hillel, 1980b). It can usually be estimated reasonably well, however, by the volumetric water content value on the desorption curve that corresponds to a pressure head of -1500 kPa (i.e., $\psi_{PWP} = -1500$ kPa), notwithstanding that crop extraction of soil water below this water content has been observed occasionally (Cutforth et al., 1991; Topp et al., 1994). The FC and PWP water contents are indicated in Figure 2.1, and their importance and utility are discussed below in the section on critical soil water parameters.

Adequate soil water storage is not the only requirement for good crop production and the existence of essential root-zone organisms. The soil must also be able to transmit rapidly the needed water into the root zone, and excess water out of the root zone. This attribute is determined by the soil water transmission relationship.

B. Water transmission

The transmission or flow of water through soil is most completely described by the Richards equation (Hillel, 1980a), which for 1-D vertical flow is given by:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left[K(\theta)\frac{\partial H}{\partial z}\right] \tag{4}$$

where t is time, z is depth in the soil, $H = \psi + z$ is the hydraulic head, θ is volumetric water content, and $K(\theta)$ is the hydraulic conductivity versus volumetric water content relationship. Equation 4 implies that soil water flows in the direction of decreasing hydraulic head, H, and that the rate of flow is determined by the magnitude of the hydraulic head gradient, $\partial H/\partial z$, and the hydraulic conductivity function, $K(\theta)$. The $K(\theta)$ function is the soil's water transmission relationship, and it gives the permeability of the soil to water as a function of the volumetric water content, θ. In effect, $K(\theta)$ is a special case of Equation 2 for the transmission of water.

$K(\theta)$ depends strongly on the magnitude and shape of $\theta(\psi)$, and $K(\theta)$ is not a single value, but a relationship where K deceases as θ decreases. Through its connection with $\theta(\psi)$, $K(\theta)$ depends on the number and size distribution of the soil pores, which in turn depend on soil porosity, structure, texture, organic matter content, and clay mineralogy. Unlike $\theta(\psi)$, however, $K(\theta)$ is also highly dependent on pore morphology parameters such as pore tortuosity, pore roughness, pore connectivity, and pore continuity. These additional dependencies can cause K to change by a million-fold or more between the saturated and permanent wilting point water contents. They also cause $K(\theta)$ to be extremely variable both within and among soil types.

Owing to the extreme sensitivity of K to pore size and morphology, the magnitude and shape of the $K(\theta)$ relationship can be altered substantially by soil structure. Example $K(\theta)$ relationships for unstructured and structured soils are given in Figure 2.2, in which it is seen that when the soil is structured the near-saturated K values can change very rapidly with changes in θ and K can be several orders of magnitude greater when the soil is structured compared to when it is unstructured. Because of this, it is not uncommon for well structured loam and clay soils to have saturated hydraulic conductivity values (i.e., K at $\theta = \theta_s$, $\psi = 0$) that are close to the saturated hydraulic conductivities of sandy soils (Fig. 2.2).

Because the $K(\theta)$ relationship determines the permeability of the soil to water, it is the primary soil property affecting the rate at which water can infiltrate the top of the crop root zone, drain out the bottom of the root zone, and redistribute within the root zone. The $K(\theta)$ relationship must therefore be optimized for maximum crop production and ecosystem health. For example, inadequate infiltration and redistribution rates can cause insufficient supply of water to crop roots and soil organisms, as well as excessive surface runoff and soil erosion. Insufficient drainage rates can cause poor soil aeration (waterlogging) in the root zone. Excessive drainage rates, on the other hand, can cause substantial loss of water, nutrients, and pesticides out the bottom of the root zone, reducing their availability to crops and soil organisms, and increasing the potential for nutrient and pesticide contamination of groundwater resources.

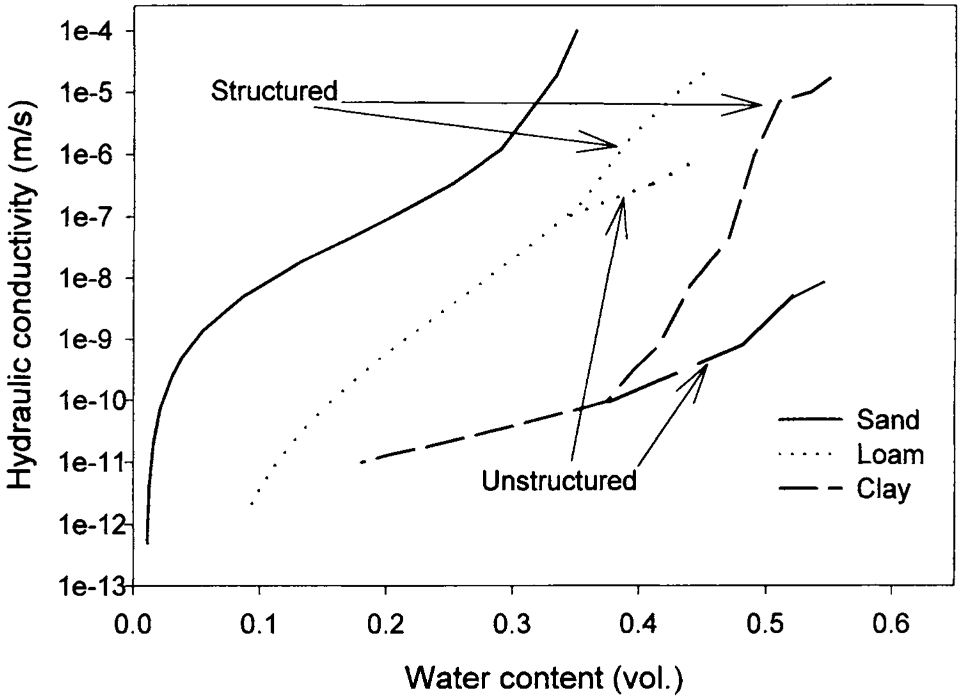

Fig. 2.2. Examples of hydraulic conductivity relationships for different soil textures and for unstructured and structured soil conditions.

C. Critical soil water parameters

As mentioned above, maximum crop production and ecosystem health require an optimization of the volumetric water content, $\theta(\psi)$, and hydraulic conductivity, $K(\theta)$, relationships. The means by which the soil and/or crop might be manipulated to achieve this is still poorly and incompletely understood. At this point we must rely primarily on a number of indirect or incomplete indicators of how well a soil stores and transmits water. These indicators take the form of critical and/or limiting soil water storage and transmission parameters, which are largely relative rather than absolute. They are discussed below and summarized in Table 2.1.

1. Critical water contents

The FC and PWP water contents can be used directly as critical soil water parameters. Many field crops perform best when the soil water content is maintained at or slightly below the FC value, provided that soil aeration is also adequate (Heermann et al., 1990). The performance of most crops is substantially impaired, however, well before the water content is reduced to the PWP value (ibid.). Consequently, FC and PWP are often used as guidelines for irrigation scheduling. In humid climates, irrigation should usually proceed only until the FC value is reached as additional water reduces soil aeration and also promotes loss of water, nutrients,

TABLE 2.1

Selected soil water parameters, their critical or limiting values, and some comments on their main application

Parameter[1]	Critical limit or function	Remarks
FC	Irrigation limit, drainable porosity, aeration limit	Widely used in water management in humid regions
PWP, AWC	Plant water supply, irrigation scheduling, soil water budget models	Does not account for water transmission
K_s	Land suitability, drain spacing requirements	Highly variable parameter and easily affected by soil management
K_{FC}	Low intensity irrigation	Allows better assessment and management of leaching losses of water and agrochemicals
R	Water conductiveness	New parameter for characterizing soil water transmission

[1]FC field capacity, PWP permanent wilting point, AWC available water capacity, K_s saturated hydraulic conductivity, K_{FC} hydraulic conductivity at field capacity, R flow-weighted mean pore size.

and pesticides through drainage below the root zone (Fig. 2.1). In dry climates, it is advisable to raise the water content above the FC periodically, so that the ensuing drainage will flush out phytotoxic salts, which tend to accumulate in the root zone because of high water loss through evapotranspiration. In general, irrigation should commence well before the root zone soil water content declines to the PWP value (discussed further below). A detailed treatment of the science and art of irrigation scheduling is given in Stewart and Nielsen (1990).

2. Critical pressure heads

Sometimes the FC and PWP pressure heads, ψ_{FC} and ψ_{PMP}, respectively, are used to schedule irrigation rather than using the corresponding water contents (ibid.). Many irrigation specialists maintain that plant water stress is controlled more by the energy status of the soil water (as indicated by pore water pressure head, ψ) than by the actual amount of soil water present (as indicated by volumetric water content, θ). For example, it is often recommended that corn, wheat, alfalfa, sunflower, and sorghum be irrigated when the root zone ψ declines to about −200 kPa, whereas peanut and potato should be irrigated at $\psi = -20$ to -60 kPa (ibid.).

3. Available water capacity

One of the most long-standing and widely used water storage parameters is the plant available water capacity, AWC (Cassel and Nielsen, 1986). This parameter is defined as the difference between the field capacity (FC) and permanent wilting point

(PWP) water contents (i.e., AWC = FC − PWP), and it estimates the amount of stored soil water in the crop root zone that is available for crop use. Because AWC is based on two points on the $\theta(\psi)$ relationship, it is controlled, like $\theta(\psi)$, by soil porosity, structure, texture, organic matter content, and clay mineralogy. One advantage of the AWC parameter is that it is a single, intuitive value that is easily measured and can be calibrated to the rooting depths of particular crops. Disadvantages, however, are that it does not account for how the soil releases its stored water, as determined by the shape of the water retention relationship, $\theta(\psi)$; it does not incorporate the water transmission relationship, $K(\theta)$; and it often changes during the growing season as crop roots grow into deeper soil layers. Another important disadvantage is that the optimum water content for the growth of many crops does not extend over the entire AWC range, because (as mentioned above) most crops become water-stressed at water contents that are greater than the PWP value. As a result, crop growth and/or performance can be significantly impaired even though substantial AWC is still available in the root zone (Stewart and Nielsen, 1990). Nevertheless, AWC has been used quite effectively in soil water budget and crop growth models for developing land and crop management recommendations. Decisions on whether to fallow or plant a crop, fertilization rate, and type of crop to plant have been based on model predictions of amount of stored water at seeding time (Akinremi and McGinn, 1996; Ash et al., 1992; de Jong and Bootsma, 1996; O'Brien, 1992; Raddatz, 1992; Raddatz et al., 1996). Such broad-scale models have also been applied to predict crop yields, and set levels of crop insurance compensation (de Jong and Bootsma, 1996; O'Brien, 1992). The AWC parameter is also used extensively as a criterion for adding irrigation water. It is often recommended that irrigation commence when the amount of water in the root zone has been depleted to about 50% of the AWC value (Gordon et al., 1996; Heermann et al., 1990). This level of depletion is sometimes referred to as the "Readily Available Water Capacity", or RAWC (Hillel, 1980b).

Improving the AWC or RAWC of a soil to increase crop production involves adjusting the soil's FC and/or PWP values. Purely from the perspective of water supply, FC should be as large as possible and PWP as small as possible in order to maximize the AWC or RAWC. The FC value should not, however, be greater than about 80% of the total soil porosity, ε, otherwise inadequate soil aeration could result (see Section IV). Because FC is defined as the water content at the cessation of gravity drainage from saturation, it is strongly affected by the large pores in the soil and can consequently be adjusted (within limits) by manipulating the soil's structure and organic matter levels (ibid.). The PWP is much more difficult to manipulate, because at $\psi_{PMP} = -1500$ kPa, soil water resides in the finest micropores only, the larger matrix and structural pores being drained and air-filled. The PWP is consequently controlled primarily by factors such as soil texture and humus content, which are difficult to adjust. As a general guideline, FC is on the order of 3–10% in sandy soils, about 10–25% in loamy soils, and about 25–50% in clayey soils; the PWP is usually on the order of 1–5% in sandy soils, 5–15% in loamy soils, and 15–20% in clayey soils (Hausenbuiller, 1978). The AWC consequently ranges from about 2–8% in sandy soils, 5–15% in loamy soils, and 10–30% in clayey soils (the

RAWC is half these values). Note that the above values are only representative "ball park" figures, and individual soils may have considerably different FC, PWP, and AWC/RAWC values as a result of specific soil structural and organic matter conditions.

4. Drainable porosity or specific yield

Another critical water storage parameter is the drainable porosity (DP) or specific yield (SY), which is often defined as the difference between the total soil porosity, ε, and the field capacity water content, FC (i.e., SY = DP = ε − FC). This parameter has been used in the design of subsurface drainage systems, in which it is a criterion for determining the depth and spacing of the drainage tubes (Bear, 1979). The SY is also numerically equivalent to the air-filled porosity or field air capacity, FAC, which is the volumetric air content of the soil at the FC water content. The FAC is a critical soil aeration parameter (Section IV).

5. Saturated hydraulic conductivity

The first and most widely used soil water transmission parameter is the saturated hydraulic conductivity, K_s, which is the permeability of the soil to water at the saturated water content, θ_s (i.e., the maximum K values in Fig. 2.2). This value is the main soil parameter determining the rate at which excess soil water drains below or out of the crop root zone, and as a result, K_s is related to the possible agricultural uses for soil in humid climates (Marshall and Holmes, 1988). Soils within the optimum K_s range (Table 2.2) tend to allow adequate infiltration and redistribution, sufficiently rapid drainage of excess water, and limited drainage of plant available water. Soils below the optimum range, on the other hand, tend to have inadequate aeration for most crops, whereas those above the range tend to have insufficient

TABLE 2.2

Saturated soil hydraulic conductivity, K_s, and optimum agricultural use (modified from Marshall and Holmes, 1988)

Range in K_s (ms^{-1})	Soils associated with K_s range	Optimum agricultural use
$<10^{-7}$	Soils of fine texture and poorly aggregated soils of medium texture. Mottling of subsoils common in humid areas	Drainage may be so slow as to restrict use of land to shallow-rooted crops. Restricted drainage is required for flooded rice
10^{-7} to 10^{-4}	Mainly sandy and loamy soils, also includes clay soils that have extensive and stable structure	Preferred soils for most crops, including those under irrigation
$>10^{-4}$	Soils of very coarse texture	This high conductivity is usually associated with poor retention of water. Use may be restricted to deep-rooted and drought-tolerant crops

plant available water (Table 2.2). Drainage engineers use K_s in the design of subsurface drainage systems, in which it is a critical parameter in determining the required spacing and size of the drainage tubes (Bear, 1979). The K_s value has also been used extensively as a critical parameter for indicating changes in soil structural quality as a result of changes in crop and/or land management practices. As indicated above, K_s is highly sensitive to changes in pore size, roughness, tortuosity, and connectivity (Hillel, 1980a). Thus even modest changes in pore morphology induced by cropping or land management can result in large changes in K_s, which in turn indicate substantial changes in the rates at which water is transmitted through the soil. For example, Ankeny et al. (1990) found that modest wheel traffic compaction by farm implements caused a reduction in K_s by one to two orders of magnitude on a silty clay loam soil under no-tillage and chisel plough tillage.

6. Hydraulic conductivity at field capacity

The hydraulic conductivity under moist (but unsaturated) soil conditions, such as the conductivity at field capacity (K_{FC}), is becoming an important soil parameter for the design and operation of the more recently developed low-intensity, high-frequency irrigation techniques. These new techniques, such as low-intensity sprinkling, sub-irrigation, and drip or trickle irrigation (Kruse et al., 1990), greatly increase the ability to establish and continuously maintain soil moisture at optimum crop production levels, which (as mentioned above) generally occur at or just below the field capacity water content, FC (Hillel, 1980b). Under such conditions, detailed knowledge of the magnitude and spatial distribution of K_{FC} can greatly assist in balancing the rate, amount, and frequency of irrigation against water use by the crop and loss of water, nutrients, and/or pesticides via drainage below the root zone.

7. Water content and hydraulic conductivity of soil macropores and matrix pores

Soil water storage and transmission parameters that are currently under development include the saturated water content and hydraulic conductivity of the soil matrix pores and the soil macropores. Matrix pores are defined as all pores that are small enough to remain water-filled at a specified pore water pressure head, ψ_m, whereas macropores (e.g., large cracks, worm holes, root channels, large interaggregate spaces, etc.) are pores that are too large to remain water-filled at ψ_m. The value of ψ_m is at present not fixed, but values of -0.3, -0.6, and -1.0 kPa have been used (e.g., Watson and Luxmoore, 1986; Timlin et al., 1994; Reynolds et al., 1997). The saturated volumetric water content of the soil matrix, θ_m, therefore corresponds to θ at $\psi = \psi_m$ on the $\theta(\psi)$ relationship (Fig. 2.1); the saturated hydraulic conductivity of the soil matrix, K_m, corresponds to K at $\theta = \theta_m$ on the $K(\theta)$ relationship (Fig. 2.2). For the soil macropores, the saturated water content or "macroporosity", θ_p, is given by:

$$\theta_p = \theta_s - \theta_m \tag{5}$$

and the macropore saturated hydraulic conductivity, K_p, by (Watson and Luxmoore, 1986; Timlin et al., 1994):

$$K_p = K_s - K_m \tag{6}$$

where θ_s and K_s are the saturated volumetric water content and saturated hydraulic conductivity, respectively, of the whole soil. Given that ψ_m is very close to zero and the soil is nearly saturated, macropores tend to be large in size (equivalent diameters of 0.3 to 1mm) but small in total volume relative to the matrix pores (Beven and Germann, 1982). Consequently, θ_p is usually small relative to θ_m, while K_p is often one to four orders of magnitude greater than K_m (ibid.). The term macropore as used here represents pores having equivalent diameters an order of magnitude larger than defined later in relation to aeration porosity.

8. Flow-weighted mean pore size

Another soil water transmission parameter that is currently under development is the "flow-weighted mean pore size", R, which is given by (Philip, 1987):

$$R = \frac{\sigma K_0}{\rho g M_0} \tag{7}$$

where σ is the air–water interfacial tension, ρ is the density of water, g is the gravitational acceleration constant, and K_0 and M_0 are hydraulic conductivity and matric flux potential, respectively, for a specified pressure head, ψ_0. The R value has been referred to as an effective "equivalent mean" pore radius that conducts water when infiltration occurs at $\psi = \psi_0$ (White and Sully, 1987). It has also been referred to as an index parameter (rather than a physical pore radius) that represents the mean "water-conductiveness" of the hydraulically active soil pores (Reynolds et al., 1997). The term water-conductiveness was coined in recognition that since R is derived from water flow (via Eq. 7), it must therefore represent in some fashion the combined size, tortuosity, roughness, and connectivity of the water-conducting soil pores (Reynolds et al., 1997). As with θ and K, the R value can be determined for soil matrix pores (R_m) and soil macropores (R_p) by substituting the appropriate K and M values into Equation 7.

The θ_m , θ_p, K_m, K_p, R_m, and R_p values have so far been used primarily to assess the impact of cropping and land management practices on soil water storage and transmission properties. For example, Table 2.3 compares these parameters for a loam soil under long-term native forest, no-tillage, and conventional moldboard plough tillage. It is seen that θ_p comprises only 3–10% of the total pore space, but R_p is one to two orders of magnitude greater than R_m and K_p, is two to three orders of magnitude greater than K_m. The parameters also show how the different land management practices have caused substantial changes in the way water is stored and transmitted in this loam soil.

IV. SOIL AERATION

Soil aeration refers to the ability of the soil to store and transmit air, particularly the common atmospheric gases, oxygen (O_2) and carbon dioxide (CO_2). Continuous absorption of O_2 and release of CO_2 by plant roots (i.e., root respiration) are fundamental requirements for crop growth and production. An adequate volume

TABLE 2.3

Some soil water storage and transmission parameters calculated from data in Reynolds et al. (1997)

Parameter[1]	Unit	Native Forest	No-Till	Conventional Till
θ_m	$m^3\ m^{-3}$	0.52a[2,3]	0.45b	0.39c
θ_p	$m^3\ m^{-3}$	0.09a	0.05b	0.03c
K_m	$m\ s^{-1}$	3.2E-7a	1.3E-7b	3.4E-7a
K_p	$m\ s^{-1}$	3.2E-4a	4.6E-5b	2.3E-5b
R_m	mm	0.077a	0.047b	0.050b
R_p	mm	5.3a	1.6b	0.61c

[1]θ_m water content when matrix pores remain filled, θ_p portion of the saturated water content held in macropores, K_m matrix saturated hydraulic conductivity, K_p macropore. saturated hydraulic conductivity, R_m flow weighted mean pore size for matrix pores, R_p flow weighted mean pore size for macropores.

[2]Values represent treatment means (10 replicates per treatment).

[3]Values within a row followed by the same letter are not significantly different at $P < 0.05$.

and continuous supply of O_2 is also required by many root zone soil organisms (e.g., earthworms and microorganisms) for their own life, and for their consequential decomposition of soil organic matter into essential crop nutrients. If O_2 is lacking in the soil or CO_2 levels are excessive, plant roots are inhibited in both their growth and their ability to absorb nutrients and water. A lack of O_2 may also result in anaerobic decomposition in the soil and an associated build up of certain gases (e.g., methane and ethylene) and compounds (e.g., manganese and sulphide precipitates) that can be toxic to both plants and soil organisms. Proper soil aeration is therefore essential for good crop production and overall soil health. The soil must be capable of adequate storage and transmission of atmospheric O_2 into the root zone, and of adequate transmission of potentially harmful gaseous by-products out of the root zone.

A. Air storage

The volume of air present or "stored" in a soil, the volumetric air content, ε_a, is $\varepsilon - \theta$. The soil air is that portion of the porosity not occupied by water. Thus ε_a is controlled by soil porosity and volumetric water content, with ε_a decreasing with decreasing ε and increasing θ. As shown in Section III, θ is a highly variable quantity that depends on soil texture, structure, and pore water pressure head. ε_a is therefore also highly variable and dependent on these parameters. As such, it is possible to specify a "field air capacity" (FAC), defined as the volume fraction of air present in the soil at the field capacity water content (FC). The FAC is consequently a measure of how much air is stored in the soil under freely drained field conditions.

FAC depends primarily on soil texture and structure or aggregation. In sandy soils it is on the order of 25% or more, in loamy soils it is generally 15–20%, and in clayey soils it is often below 10%. Highly structured or aggregated soils, particularly fine-textured ones, can have an FAC of 20–30%, because these soils contain many large pores that are almost always air-filled (i.e., water drains quickly from the large pores at high pore water pressure heads). Compacted, structureless soils, on the other hand, can have an FAC below 5%.

Although the connection between volumetric air content, FAC, and soil aeration is obvious, it is unfortunately neither direct nor predictable. Reported field air capacity values at which soil aeration is limiting to root respiration (and hence crop growth) vary from 5% to 20%, which is almost the entire FAC range. One reason for this is that soil aeration depends not only on the volume fraction of air in the soil, ε_a, but also on the rate of exchange of air between the soil and the atmosphere (i.e., the ability of the soil to transmit air).

B. Air transmission

Advection and diffusion are the two main mechanisms by which gas is transmitted through soil. Advection is the mass flow of air in and out of the soil. Diffusion is the movement through soil of individual gas components of air (e.g., O_2 and CO_2) in response to gradients in the concentrations of the gas components.

Advection is caused primarily by changes in total air pressure between the soil air and the atmosphere; water infiltration and drainage; the rise and fall of a shallow water table; and extraction of soil water by crop roots. Changes in total air pressure result from atmospheric changes in temperature and barometric pressure (Romell, 1922) and from wind gusts over the soil surface (Scotter et al., 1967; Kimball and Lemon, 1971). Increasing soil water content, in response to infiltrating rainfall, irrigation water, or the rise of a shallow water table, causes the advective flow of soil air into and out of the soil in response to an advancing infiltration or wetting front. Decreasing soil water content resulting from soil water drainage, soil water extraction by crop roots, or the fall of a shallow water table, causes the advective flow of atmospheric air into the soil (Romell, 1922; Lance et al., 1973).

Because air pressure gradients in the soil are generally small, the permeability of the pore space to air, $k(\varepsilon_a)$, is the dominant parameter affecting advection. The $k(\varepsilon_a)$ function is not a single value, but a relationship where k decreases in an exponential-like fashion as ε_a decreases. Obviously, $k(\varepsilon_a)$ is generally less in wet soils than it is in dry soils. Owing to the connection between ε_a and θ, $k(\varepsilon_a)$ depends on the number, size distribution, and interconnection of the air-filled soil pores. The extreme sensitivity of $k(\varepsilon_a)$ to the size and morphology of the air-filled pores means that the magnitude and shape of $k(\varepsilon_a)$ can be altered substantially by changes in soil structure.

Gas diffusion through soil occurs because soil biological respiration causes the concentrations of the gases in the soil to differ from those of the atmosphere. For example, O_2 diffuses into the soil because its consumption by plant roots and soil organisms reduces the O_2 concentration in the soil air below the O_2 concentration in the atmosphere. Similarly, CO_2 diffuses out of the soil because its production by roots and organisms raises the CO_2 concentration in the soil air above that of the atmosphere.

Gas diffusion through soil is often described by the relationship (Cook, 1995; Jury et al., 1991):

$$\frac{\partial C_g}{\partial t} = \frac{\partial}{\partial z}\left[D_g \frac{\partial C_g}{\partial z}\right] \pm S_g \tag{8}$$

where C_g is the concentration of gas in the soil air, t is time, z is depth in the soil, D_g is the effective gas diffusion coefficient in the soil, and S_g is a source/sink term describing the rate of consumption or production of the gas within the soil. The gas concentration in the soil (C_g) is determined by the effective gas diffusion coefficient (D_g), the gas concentration gradient ($\partial C_g/\partial z$), and the rate of gas production or consumption in the soil (S_g). The D_g value depends directly on soil porosity and volumetric air content (Millington and Quirk, 1961; Sallam et al., 1984), and indirectly on the morphology and interconnection of the soil pores.

The relative importance of soil gas advection versus diffusion as mechanisms for overall soil aeration is a subject of some debate. It is usually assumed that diffusion is the primary mechanism because (unlike advection) it occurs almost continuously over a range of depths in the soil. The transmission parameters controlling soil aeration in the crop root zone are the air permeability relationship, $k(\varepsilon_a)$, the effective soil gas diffusion coefficient, D_g, and the soil gas consumption/production rate, S_g.

C. Critical aeration parameters

The aeration requirement of the soil is determined by the amounts of soil gases consumed (e.g., O_2) and produced (e.g., CO_2) under optimum conditions (i.e., no limitations) by all soil biological activity (both flora and fauna). The aeration requirement depends on crop type and its stage of development, soil type and structure, temperature, soil water content, organic matter content, and the activity and life cycles of macro- and micro-organisms. The ability of the soil to satisfy the aeration requirement depends primarily on the interaction of the field air capacity, air permeability, air diffusion coefficient, and the rate of gas consumption/production. Indicators of overall soil aeration status must therefore integrate these parameters to as great an extent as possible. Although many indicators of soil aeration status have been proposed (Glinski and Stepniewski, 1985), the most commonly used include air-filled porosity, air permeability, soil air composition, soil respiration, and oxygen diffusion rate. These indicators are discussed below and summarized in Table 2.4.

1. Air-filled porosity and air permeability

One of the earliest measurements of aeration was the determination of the volumetric air content, or air-filled porosity (ε_a), at some standardized value of soil water content or matric potential. This was done by calculating ε_a for an undisturbed sample of field soil that was either already at field capacity, or was saturated in the laboratory and then drained, using a tension table, to a specified matric potential, usually the FC value. The resulting air-filled porosity is the field air capacity, FAC. The air-filled pores at FAC will generally have equivalent diameters of $> \approx 50$ μm. The available evidence suggests that $\varepsilon_a = 0.1$ (i.e., 10% of the total soil volume filled

TABLE 2.4

Selected soil aeration parameters, their critical or limiting values, and some comments on their application

Parameter	Function/ Limit	Remarks
Air-filled porosity, ε_a	Adequate soil aeration usually requires $\varepsilon_a > 0.1$	Static air storage parameter, does not consider soil air advection or diffusion
Air permeability, $k(\varepsilon_a)$	Indirect indicator of soil aeration	Does not consider air diffusion or storage
Soil oxygen concentration	Adequate soil aeration usually requires >10 g m^{-3}	Static parameter but may include the combined effects of air storage, advection and diffusion
Soil respiration rate	Indirect indicator of soil aeration as respiration rate is partially dependent on amount of O_2 and CO_2 present	Distribution with depth is difficult to obtain because of cumbersome equipment
Oxygen diffusion rate, ODR	On average, crop root growth requires ODR >12 μg m^{-2} s^{-1}	Limiting ODR values depend on crop, soil temperature and other factors

with air) is the lower limit for adequate aeration of the deepest roots of most common field crops (Wesseling and van Wijk, 1957). Although the air-filled porosity obviously gives a direct measure of the amount of air "stored" in the soil, it is probably not a good overall indicator of soil aeration because it does not account for the dynamics of soil aeration (i.e., advection and diffusion).

Direct measurement of air permeability, $k(\varepsilon_a)$, has also been used to indicate soil aeration. This measurement provides information on the effective sizes and continuity of the air-filled pores, which is crucial to adequate soil aeration (Ball et al., 1988; Grant and Groenevelt, 1993). The $k(\varepsilon_a)$ measurement lacks information on air storage and diffusion, however, and is therefore probably not a good overall indicator of soil aeration.

2. Composition of soil air

Measurement of the composition of the soil air, particularly the O_2 and CO_2 concentrations, have been used extensively as indicators of soil aeration. The concentrations of O_2 can be measured directly and *in situ* using electrodes placed inside hollow access tubes inserted into the soil (Willey and Tanner, 1963; Letey and Stolzy, 1964; Willey, 1974). The volume of air surrounding the electrodes must be equilibrated with the soil air prior to measurement, however, which may require many hours or even several days in wet soils. In another approach, small syringe samples of soil air ($\approx$0.5 ml) are injected into a gas chromatograph to obtain both the overall soil air composition and the concentration of selected gas components. A third, indirect, method of inferring soil O_2 concentration is to measure the soil's oxidation–reduction (redox) potential using a platinum-reference electrode couple

(Linebarger et al., 1975; Flühler et al., 1976). The method measures the overall oxidation state (in millivolts, mV) of the soil, which is indirectly related to soil O_2 concentration. Although the redox method is straightforward and easy to apply, it has many problems (Patrick et al., 1973; Patrick and Henderson, 1981), and the results are often ambiguous. Carter (1980) found that the critical redox potential below which sugar cane yield was affected was 332 mV, and the critical value for crop emergence appears to be about 440 mV (Glinski and Stepniewski, 1985). Meyer and Barrs (1991) found, for a range of plants, that root growth stopped when the O_2 concentration in the soil fell below 10 g m^{-3}. Although soil air composition is essentially a static measurement, it may still be a plausible indicator of soil aeration, because gas concentration at any given position or time will be the result of the combined, albeit largely unknown, effects of air storage and transmission.

3. Soil respiration

As soil respiration (i.e., the rate of consumption of soil O_2 and production of CO_2 in the soil) is clearly linked to soil aeration (e.g., the S_g term in Eq. 8), measurement of respiration will give an indication of soil aeration. Soil respiration can be measured either in situ or in the laboratory. Laboratory methods, described in Orchard and Cook (1983) and West and Sparling (1986), involve measurement of the rate of CO_2 accumulation in a sealed flask under controlled conditions and give a measure of biological activity. Field methods use either profile measurements (de Jong and Schappert, 1972; Magnusson, 1989; Barber et al., 1990) or chamber methods. The profile methods consist of calculating differences between diffusion into and out of each layer of soil to determine the rate of production/consumption of gas within the layer, whereas the chamber methods determine the rate of CO_2 flux at the soil surface as an integrated measure for respiration for the total soil depth. The chamber methods are either static (Ross and Tate, 1993; Lundegårdh, 1927; Witkamp, 1969), in which CO_2 is absorbed by an alkali solution, or dynamic, in which the increase in CO_2 concentration is measured as a function of time (Parkinson, 1981; Rochette et al., 1992; Norman et al., 1992). The chamber methods only provide a surface flux of CO_2, and consequently the measured respiration by this method is only "apparent" because its source distribution within the soil profile is unknown. However, Witkamp and Frank (1969) used a chamber method to measure CO_2 flux as soil layers were progressively removed, allowing the average respiration for each layer to be estimated.

4. Oxygen diffusion rate

The oxygen diffusion rate (ODR) is a measure of the flux of O_2 to a root-like, O_2-reducing platinum electrode that is maintained at a constant electrical potential, usually −0.65 volts (McIntyre, 1970). The steady current measured by the electrode corresponds to the rate at which O_2 is transmitted through the soil to the probe, where it is electrochemically reduced to water (H_2O) or hydroxyl ions (OH^{-1}). It is assumed in this measurement that the root-sized probe approximates root respiration (i.e., O_2 consumption by the root). Although the measurement is referred to as a "diffusion rate", it is actually a measure of the combined effects of advection and diffusion since it measures the total flux of O_2 to the probe. It is theoretically possible with this method to calculate the limiting ε_a depth profile for adequate aeration in a soil, but this requires several assumptions and approximations (Glinski and Stepniewski, 1985). Letey and Stolzy (1964) quote an ODR value of 67 $\mu g\ m^{-2}\ s^{-1}$ as the limiting value below which soil aeration is limiting to crop growth. Blackwell and Wells (1983) found, however, that oat growth did not slow until the ODR dropped below 9.3 $\mu g\ m^{-2}\ s^{-1}$, and did not stop altogether until the ODR was effectively zero. Glinski and Stepniewski (1985) suggest a limiting ODR of 12 $\mu g\ m^{-2}\ s^{-1}$ as a working compromise. The ODR value is potentially a good indicator of soil aeration because it combines the effects of $k(\varepsilon_a)$, D_g, and S_g, and because depth profiling of the limiting ε_a is at least theoretically possible. As indicated above, however, decisive limiting ODR values are yet to be defined, and the ODR also appears to be dependent on soil temperature and the size and shape of the platinum electrode. The effects of soil air storage (air-filled porosity) are also not directly accounted for by this measurement.

V. SOIL STRENGTH

Optimum crop production and good overall ecosystem health depends critically on soils having a range of strengths that provide a balance among the requirements for traffic, crop growth, and other soil processes. Soil strength relates to the soil's mechanical responses to the forces associated with crop production activities and processes. A soil with desirable strength resists the loss of its structure through tillage and traffic and holds plants upright throughout the growing season. At the same time the soil must be weak enough to permit seedling emergence, root penetration, and burrowing soil fauna. Thus the optimum soil strength is a balance between conflicting needs. As soil strength varies with water content, some of the needs, such as for trafficking, are achieved by careful timing of operations in relation to soil conditions.

A. Nature and origins of soil strength

The main components of soil strength are shear strength, compressive strength, and tensile strength (Hillel, 1980a). Shear strength is the ability of the soil to resist lateral or sideways sliding movement under a tangential force, such as might occur at the sides of a tillage tool or under a wheel that is slipping or spinning. Shear strength arises primarily from the friction associated with the sliding of irregularly shaped soil particles past each other, and their cohesiveness. Compressive strength is the ability of the soil to resist compaction (i.e., reduction in total porosity) due to forces directed inwardly. Tensile strength is essentially the reverse—the ability to resist expansion (increase in total porosity) due to outward acting forces. Inward or compressive forces occur directly in front of most tillage tools and directly under wheels, whereas outward or tensile forces usually occur on the lifting edges of tillage tools. Both compressive and tensile strength (and to some extent shear strength) arise from how tightly packed together the soil particles are, which is exhibited as a dependence on soil density. All components of strength arise from electrostatic attraction (cohesion) between clay minerals, the presence of organic and mineral binding agents within and between soil particles, and adhesion between soil particles caused by the contractile force of pore water at subatmospheric pressures. Each of these factors is affected by changing water content. The last factor (pore water induced adhesion) is the most highly variable because of the large variation in soil water or matric potential that normally occurs in space and time with the varying water contents experienced by crops (Greacen, 1960; Petersen, 1993). All three components of soil strength depend on soil water potential and should therefore be referenced to the water content of the soil (or even better, to soil water potential) at the time of their measurement.

B. Measures of soil strength

There are both fundamental and empirical measures of soil strength. Fundamental measures are preferred because they yield true strength properties of the soil. Empirical measures, on the other hand, yield only "apparent" strength properties, which are usually inextricably confounded mixtures of true strength properties. In addition, empirical measures of soil strength tend to be highly dependent on the technique or apparatus used to make the measurement.

Despite the above-mentioned limitations, empirical measures of soil strength have traditionally been preferred in agriculture over fundamental measures. This is because empirical measures of soil strength tend to be simpler and field-based, and they often yield more intuitive reflections of what plants and soil actually experience during agricultural activities than do fundamental measures. Fundamental measures have been mainly applied in predicting soil behaviour in response to applied stresses during traffic or tillage or other soil processes. The complexity of soil strength and the response of soil to stresses has meant that many applications of soil deformation theory are simplified and made specific to the intended use, such as rutting depth by wheels and the design and performance of tillage tools. Critical state soil mechanics offers a unifying framework now being actively applied to unsaturated agricultural soils to predict soil strength, deformation, and structural rearrangement (Bailey and Vandenberg, 1968; Hettiaratchi and O'Callaghan, 1980; Kirby, 1991a, 1994; Petersen, 1993). Future advances in the characterization and management of agricultural soil strength will come mainly through the use of fundamental strength measurements, rather than empirical measurements.

C. Critical soil strength parameters

Important fundamental measures of soil strength include pre-consolidation stress, cohesion, internal friction, and critical state properties. The main empirical measures of soil strength include cone penetration, vane/ring shear, plate bearing tests, and plastic limit. Both fundamental and empirical measures are described below and summarized in Table 2.5. In general, the existing critical agricultural soil strength parameters are few in number and consist of essentially limiting values for tillage,

TABLE 2.5

Selected soil strength parameters, their function with critical values and comments on their use

Parameter(s)	Function/ critical value	Comments
Fundamental properties		
Preconsolidation stress, P_c	Limit to resist compaction / >100 kPa	Easy to measure, useful limit
Cohesion and friction	Strength limit to resist shear breakage	Used in tillage modelling and design
Critical state properties	Deformation response to shear and compaction stresses	Used in tillage and compaction modelling
Empirical properties		
Cone penetration resistance	Root growth limit/ < 2 MPa	Easy to measure, popular
Plate bearing tests	Trafficability limits	Correlates to sinkage beneath tires and tracks
Ring shear/vane shear	Limit to resist shear breakage	Used in compaction and tillage studies
Plastic limit	Compaction and workability limits	Widely used empirical measure

trafficking, root penetration, and crop planting. These limits are based on pre-consolidation stress, cone penetration resistance, and plastic limit.

1. Pre-consolidation stress

Pre-consolidation stress (P_C) is the maximum pressure (force per unit area) that has acted on the soil at some time in the past. In the plow layer, the relevant P_C is usually that produced by wheels and tillage implements during the last few tillages or growing season or the maximum stress resulting from soil drying. Below the plow layer (i.e., the B and/or C horizons), the relevant P_C might come from the overburden weight of the A horizon, major dessication giving rise to high stresses, or, in higher latitudes, from the weight of ice sheets during the last glaciation (Veenhof and McBride, 1996; McBride and Joosse, 1996). Soil deforms in a manner dependent on the magnitude of the applied stresses (force per unit area) relative to the P_C of the soil. When the applied shear and compressive stresses are small relative to P_C, there is little deformation of the soil, such as when a light vehicle is driven over soil that has already been compacted by a heavy vehicle. When the compressive stress is small but the shear stress large relative to P_C, the soil expands, which is what usually occurs during tillage. When both the compressive and shear stresses are large relative to P_C, the soil compacts, as can occur when excessively heavy agricultural implements are used. Pre-consolidation stress is usually measured in a triaxial cell or lever arm loader (Bradford and Gupta, 1986; Kirby, 1991a.).

Pre-consolidation stress, P_C, has been used as an indicator of when a soil can be trafficked without excessive compaction. As discussed above, very little additional soil compaction occurs when an applied pressure is less than the P_C of the soil. Consequently, a soil is trafficable when its P_C is greater than the contact pressure of the wheels or tracks of the equipment that is going to be driven on the soil. As the contact pressure of the wheels and tracks of agricultural machinery is on the order of 100–200 kPa, then most soils can usually be trafficked safely when its P_C is greater than about 100 kPa (Kirby, ibid.). In the crop root zone, pore water controlled interparticle adhesion (see sub-section A above) is the main process determining when $P_C > 100$ kPa.

2. Cohesion and internal friction

Cohesion and internal friction are descriptors of shear strength. Cohesion is essentially the shear strength of a soil when there is no load or force acting perpendicular to the direction of the shearing force. Internal friction, on the other hand, is the force required to cause the soil particles to slide over each other. Cohesion arises primarily from the tendency of soil particles to stick together due to the action of organic and mineral binding agents, electrostatic attraction between clay minerals, and pore water induced adhesion. Internal friction arises primarily from the shape, roughness, angularity, and strength of the individual soil particles. Clayey soils are almost always somewhat cohesive because they contain clay minerals and organic matter. Sandy soils, which are usually uncemented and contain little organic matter, tend to be cohesive only when moist (due to pore water induced adhesion), and noncohesive when dry or saturated. Internal friction does not seem to be related to soil type, but increases strongly with soil density. Cohesion and internal friction are used extensively in the design and performance assessment of tillage tools, such as the tines on cultivators and the coulters on disks and seeders (McKyes, 1985). They have also been used to estimate the erosion potential of soil. Cohesion and internal friction are usually measured using a triaxial cell or direct shear box (Sallberg, 1965).

3. Critical state properties

Critical state properties (Bailey and Vandenberg, 1968; Hettiaratchi and O'Callaghan, 1980) have seen limited application to unsaturated soil but show great promise for predicting soil strength, deformation, and structural rearrangement in response to applied compressive and/or shear stresses (Kirby, 1991a, 1994). The critical state theory, from which the critical state properties arise, provides a unifying concept that covers the various modes of deformation (i.e., elastic, nonelastic, nonlinear) to the applied stress. These modes of deformation range in magnitude from slight during elastic phase to heavily deformed. When soils are strongly distorted by compressive or shear stresses, the relationship between volume and magnitude of applied stress becomes unique (Schofield and Wroth, 1968; Hettiaratchi, 1987). Some work suggests that critical state properties may eventually be useful for predicting the response of soils to a wide range of agricultural activities, such as tillage, trafficking, rice puddling, and mole drain installation (Kirby, 1991a, 1994). Critical state properties are measured using shear box or triaxial cell apparatus (Hettiaratchi et al., 1992; Kirby, 1991a).

4. Cone penetration

Cone penetration involves measuring the force required to push or hammer a cone-tipped rod (known as a cone penetrometer) into the soil. The measured force relates to some combination of shear and tensile failure, compression, plastic soil flow, and both metal-to-soil and soil-to-soil friction (Farrell and Greacen, 1966; Hillel, 1980a; Marshall and Holmes, 1988). A voluntary standard for the design of a portable cone penetrometer has been adopted by the American Society of Agricultural Engineers (ASAE, 1994), but many variations are in use. Cone penetration measurements have been used to study tillage effects on penetration resistance (Cassel, 1982), and to estimate soil trafficability and soil resistance to plowing, seedling emergence, and root growth (Bengough and Mullins, 1990; Bengough and Young, 1993). Cone penetration techniques and equipment most commonly used in agriculture are reviewed in Bradford (1986).

Cone penetration resistance provides an indicator of when soil strength becomes too great for effective penetration by crop roots. Extensive work has shown that for most agricultural crops root growth slows dramatically, or ceases altogether, when the cone penetration resistance exceeds about 2 MPa (e.g., Bengough and Mullins, 1990).

5. Vane/ring shear tests

The vane/ring shear tests involve measuring the torsion (rotational force) required to rotate metal plates ("vanes") or rings that are either placed on the soil surface or driven into the soil. Various techniques and apparatus are available, but a particularly popular method in agriculture involves driving a four-bladed vane into the soil, and then measuring the force required to rotate the vane in the soil (see Hillel, 1980a for schematic diagram). The force measured in a vane/ring shear test relates primarily to shear strength and has been used as an estimate of soil cohesion (Hillel, 1980a), although it is recognized that internal soil friction is present in the measurement as well (Karafiath and Nowatzki, 1978). During measurement using the Cohron sheargraph, the normal stress is both controlled and measured (Ayers, 1987; Kirby and Ayers, 1993). The resulting data can be analysed using critical state soil mechanics (Kirby and Ayers, ibid.).

6. Plate bearing tests

Plate bearing tests measure the force required to push a square plate into the soil, either to a specified depth or to the depth attained with a specified force (Karafiath and Nowatzki, 1978). The amount of sinkage of the plate for a given force has been correlated with wheel traction and soil compaction, and the test has thus been used to estimate overall soil trafficability. The results seem to be highly dependent on the size of the plate, which has not been standardized.

7. The plastic limit

The plastic limit is the minimum mass water content (mass of water per unit mass of solids) at which a soil changes from a semirigid and friable (crumbly) state to a plastic state. When soil is in a plastic state it behaves like putty (i.e., it can be moulded and rolled without breaking, cracking or crumbling; Hillel, 1980a; Craig, 1978). Plastic or putty-like behaviour requires that the soil's component particles be free to slide easily past each other, but still be strongly attracted to each other by cohesive forces. This usually requires the presence of clay minerals and organic matter, and thus plastic behaviour or "plasticity" tends to be exhibited only by fine textured clay-rich soils. The plastic limit is one of the so-called "Atterberg Limits" (e.g. Craig, 1978), and it has been used extensively to designate when a soil can be trafficked without causing excessive compaction or loss of structure (Kirby, 1988, 1990; Kirby and Blunden, 1992).

The plastic limit has been used in several ways to determine both the agricultural workability of a soil and its relation to plant available water. Most agricultural soils are not trafficable (i.e., P_C < 100 kPa) and sustain significant compaction damage when they are wetter than the plastic limit (Kirby, 1988, 1990; Kirby and Blunden, 1992). However, optimum workability (i.e., optimum condition for plowing, discing, etc.) often occurs at or just below the plastic limit (Davies et al., 1977; Dexter, 1988). Thus the plastic limit is often a key parameter determining when tillage will cause maximum benefit or maximum damage. When the plastic limit and field capacity (Section III) water contents are similar, the soil can be worked and planted soon after drainage from saturation, when the soil still contains its maximum amount of stored plant-available water. This desirable situation has been found for many of the soils in England (Davies et al., ibid.). By contrast, for many of the vertisols in Australia, Kirby (1991b) found that the plastic limit and permanent wilting point (Section III) water contents are similar (i.e., the soil cannot be worked until it has dried to the point where there is little stored water remaining for crop growth. Use of the plastic limit in combination with water storage parameters can thus indicate what crop types are feasible for the soil from a planting date and water use point of view.

VI. SOIL STRUCTURE

Soil structure has traditionally been viewed in agriculture as the combination and spatial arrangement of primary soil particles into larger secondary particles known as aggregates, peds, clods, etc. (Canadian Society of Soil Science, 1976). It has been recognized, however, that the existence and arrangement of primary and secondary soil particles also influences the size, shape, tortuosity, roughness, arrangement, and continuity of the soil pores between and within the particles. Thus, agricultural soil structure is now usually considered to refer to the spatial arrangement of both the soil particles and the soil pores within and between the particles (e.g., McKeague and Wang, 1982). In this respect, soil structure can exert a strong influence on many physical, chemical, and biological properties of the soil. From a soil physical quality perspective, the main importance of soil structure lies not in the characteristics of the structure itself, but in how the structure affects and controls the fundamental soil physical attributes of water storage and transmission (Section III), aeration (Section IV), and strength (Section V).

A. Description of soil structure

Soil structure can be organized into three broad categories (Hillel, 1982): single grain, massive, and aggregated. Single grain structure refers to a material that consists mainly of primary particles (i.e., rock and mineral fragments) that are completely unattached and randomly oriented. This structural condition often occurs in "sand" soils. Massive structure occurs when randomly oriented primary particles (e.g., clay minerals) are compressed together into large cohesive blocks, such as occasionally occurs in compacted "clay" soils. An aggregated structure refers to a mixture of attached and unattached primary and secondary soil particles, where the secondary particles consist of porous clusters of primary particles held together by various organic and inorganic binding agents. An aggregated structure is generally considered best for agricultural activities. The various soil chemical, physical, and biological processes involved with the formation of an aggregated soil structure were recently reviewed by Tisdall (1996).

The "aggregated" category of soil structure can be further differentiated on the basis of whether the aggregates were formed by natural pedogenic processes or by agricultural activities. Aggregates formed naturally (i.e., without human activity) are classified according to four morphological "ped type" categories: platy, blocky, columnar/prismatic, and granular/spherical (Canadian Society of Soil Science, 1976). These four ped types are often used to describe soil profiles, and they usually also indicate the dominant soil forming (pedogenic) processes that are operative in the soil profile. Aggregates formed by agricultural activities, on the other hand, are classified primarily on the basis of size and stability, rather than shape. This is because aggregate size and stability are the most sensitive aggregate parameters in an agricultural soil (Marshall and Holmes, 1988).

The structure of soil voids or pore space, which is a determinant of soil air–water relationships (McKeague and Wang, 1982), can also be described using concepts of shape, arrangement, and size. In addition the continuity of the pores space can also be evaluated. Pedological field studies segregate visible pores (i.e., macrostructure of pores > 200 µm) on the basis of size and shape to provide a semi-quantitative analysis (ibid.). Micromorphological methods can be used to determine directly the shape and continuity of pores < 100 µm. Pore size distribution is considered to be a good indicator of the soil structural condition (Greenland, 1979) and has proven useful for predicting water infiltration rates, water availability to plants, soil water storage capacity, and soil aeration status (Cary and Hayden, 1973; Carter and Ball, 1993).

B. Formation of soil structure

In agricultural soils, the formation of soil aggregates is dependent upon both abiotic and biotic factors, the former being mainly related to soil clay content and the capacity for natural structure forming processes (e.g., alternating shrinking and swelling, freezing and thawing, wetting and drying). When biotic factors dominate, the aggregation process often occurs in a hierarchy of three main size classes: clay micro-structures (< 2 µm diameter), microaggregates (2 to 250 µm), and macroaggregates (>250 μm) (Tisdall, 1996). Clay micro-structures, which consist primarily of clay–organic matter complexes, are formed and stabilized by humified organic matter and inorganic ions (e.g., Ca). Microaggregates are formed and stabilized by microbial materials such as polysaccharides, hyphal fragments, and bacterial cells or colonies (ibid.). Macroaggregates, which tend to be only temporarily stable, are formed primarily by plant root proliferation and the activity of soil fungi and soil fauna (ibid.). In soils (e.g., Oxisols) in which abiotic factors (e.g., Fe and Al oxides) are the main aggregating agents, the aggregation process is often random rather than hierarchical (Oades, 1993).

C. Critical soil structure parameters

Although the processes involved with soil structure formation are recognized and soil structure is commonly investigated, there are few if any accepted underlying theoretical principles. The exception to this is the aspect of soil structure that addresses pore size distribution, which has been addressed in sections III and IV. This links soil structure to the water and air storage and transmission parameters. Other soil structure parameters are dependent on the methodology and conditions used for measurements (Letey, 1991). Thus, soil structure is not easily quantified, and its description tends to be subjective.

Soil structure has both semi-permanent properties (e.g., mineralogy of primary particles), and ephemeral properties (e.g., pore and aggregate size distribution) that change with climate, season, soil wetness, and agricultural activity. In addition, many soil structure attributes tend to be both soil and site specific. This situation has so far prevented the establishment of comprehensive quantitative methodologies for characterizing soil structure; at present, only specific, incomplete and qualitative methods exist. As a result, a wide range of qualitative and overlapping critical soil structural parameters are available that are often difficult to define and interpret in a precise manner. Nevertheless, some of these parameters have been used effectively to characterize soil structural quality. Also, Kay (1990) has recently provided a useful framework for interpreting these parameters based on the concepts of soil structural "form", "resiliency", and "stability", as discussed below.

Structural form is an expression of pore space that describes the arrangement and size of the pores within and between the aggregates (e.g., size distribution and total volume of pores). Structural resiliency describes the inherent ability of a soil to regenerate its pore space arrangement after the removal of a specific stress (e.g., compaction) through natural processes. In the soil surface (i.e., plow layer), these natural processes include soil floral and faunal activity, seasonal freezing and thawing, and wetting and drying. Structural stability describes the ability of the soil to retain the distribution and size of its aggregates after exposure to external forces, particularly those of impact, shear, abrasion, and slaking. Stability can be addressed from both the pore or aggregate viewpoint, but the latter usually dominates because of its relative ease of measurement. Generally, the distribution of aggregate sizes (i.e., aggregate size distribution) and the stability of selected aggregate sizes (or size classes) are the main approaches used to assess soil structural stability. Soils that are highly stable tend to have strong, persistent aggregates, and an aggregate size distribution that includes large aggregate sizes. Sieving is the main method for obtaining the aggregate size distribution, and water is the usual reagent used to determine aggregate stability. In some cases sonification, dispersion, or shaking are used to disrupt the aggregate prior to the sieving procedure. Also, sieving may occasionally be replaced entirely by turbidimetric, densimetric, and dispersive techniques; other reagents, such as alcohol or benzene, may be used in place of water (Williams et al., 1966; Burke et al., 1986; Carter and Gregorich, 1996). Various aspects of aggregate size distribution have been used as indirect indicators of soil capacitance, and various measures of aggregate stability as indirect and qualitative indicators of soil strength. For example, the size distribution of aggregates affects the infiltration and drainage of water and soil aeration. The stability of aggregates has a large effect on the susceptibility of the soil to water and wind erosion, soil surface crust formation, hardsetting and, to some extent, compaction.

This conceptual framework allows the various morphological and largely qualitative descriptions of soil structure (e.g., aggregate size distribution) to be categorized and assessed in terms of how the soil "functions" for various purposes (e.g., crop production). From the perspective of soil physical quality, the framework allows certain descriptions of soil structure to be used and interpreted as indirect and/or relative critical parameters of soil capacitance and strength. These parameters are described below and summarized in Table 2.6.

TABLE 2.6

Selected soil structural parameters, their associated soil processes/attributes and comments on main application

Parameter	Soil processes influenced	Comment
Soil structural form		
Relative compaction	Aeration	Correlates with cereal yield
Macropore volume	Aeration and permeability	Indirect indicator of aeration
Macropore continuity	Permeability	Indirect measure of degree of interconnectedness of large pores
Soil structural resiliency		
Self-mulching index	Infiltration and workability	Involves swelling clays in aggregation
Soil structural stability		
Dry aggregates >0.84 mm	Wind erosion	Easily obtained and widely used
Water stable aggregates > 250 μm	Water erosion and crusting	Indicator of soil structural stability to crop management
Water stability of 1–2 mm aggregates	Water erosion and crusting	Stability to mechanical stresses

1. Macropore volume

Macropores are those that are air-filled at field air capacity, FAC, (see Section IV) and that have equivalent pore diameters $> \approx 50$ µm. Carter and Johnston (1989) showed that under some humid soil conditions a macropore volume below about 12% can be related to increased plant root disease. As macropores are associated with the rapid drainage of excess water and improved aeration, macropore volume (measured using intact soil cores) may potentially serve as an indirect indicator of adequate aeration in the soil root zone. At present, however, there is no agreed-upon size range for soil macropores.

2. Relative compaction

Relative compaction is the ratio of actual soil bulk density to a maximum bulk density obtained under some standard compaction treatment. It can be used as a critical parameter indicating the relative proportion of large soil pores associated with rapid drainage and aeration, to small pores which tend to retain water and exclude air. Carter (1990) found that yield of cereal grains was reduced dramatically at relative compactions < 0.90.

3. Pore size distribution

Pore size distribution is essentially the range of pore diameters that occur in the soil. It is usually obtained using soil water desorption techniques and the capillary rise equation (Hillel, 1980a). As a result, the calculated pore diameters are "equivalent" rather than actual (see Section III). Although the pore size distribution is effectively a continuous function, it is usually partitioned into a number of "size classes" where each class is deemed to have a certain property or function. For example, Greenland (1979) suggested a system of three pore size classes in which each class represented a dominant soil water process: water transmission pores (> 50 µm diam, $\psi > -6$ kPa as determined using the capillary rise equation), water storage pores (50 to 0.5 µm diam, $\psi = -6$ to -600 kPa), and residual pores (< 0.5 µm diam, $\psi < -600$ kPa). Thomasson (1978) proposed calculating the volume of "air capacity or water transmission" pores (pores > 60 µm equivalent diameter as determined using the capillary rise equation, $\psi > -5$ kPa) and the volume of "available water or water storage" pores (pores 60 to 0.2 µm equivalent diameter, $\psi = -5$ to -1600 kPa). He deemed soil structural quality to be "very good" if the volume of "air capacity" pores exceeded 15% of the total pore volume, and the volume of "available water" pores exceeded 20% of the total pore volume. Percentages of "water process pores" and "air capacity" and "available water" pores can be viewed as critical soil structural parameters that indicate the transmission and storage of water and air. In addition, the pore size limits assigned by Greenland (1979) and Thomasson (1978) match well with the matric potentials associated with the FC and PWP water contents (Section III), except for 0.5 µm ($\psi = -600$ kPa). These critical parameters are, however, only indirect and relative indicators, because the storage and transmission of water and air are not measured directly, but are instead based on the assumption (often reasonable) that larger pores serve primarily a water transmission and aeration function, while smaller pores serve primarily to store water.

4. Macropore continuity and organization

Macropore continuity may be viewed as a combined measure of the average length, tortuosity, and degree of interconnection of large soil pores (Ball et al., 1988). Macropore organization, defined as the ratio of air permeability to the air-filled porosity, represents that part of the intrinsic soil permeability that is dependent on the arrangement and shape of the macropore space, but not dependent on the total macropore volume (Blackwell et al., 1990). These parameters are being developed as indirect indicators of how soil texture (sand, silt, clay content) and agricultural activities (e.g., tillage practices, trafficking) affect the ability of soil macropores to store and transmit water and air (Carter, 1992).

5. Self-mulching index

Structural resiliency varies widely among soils, with some sandy and non-swelling clay soils demonstrating virtually no resiliency, whereas some swelling clay soils are highly resilient. Highly resilient or "self-mulching" soils are usually clay to clay loam in texture and undergo sufficient shrink–swell activity to form a friable mass of small (<5 mm) and highly stable aggregates. The self-mulching characteristic is highly desirable in agricultural soils as the aggregation tends, among other things, to produce good infiltration and drainage of water, good soil workability, and high seed bed quality (Grant et al., 1993). Grant and Blackmore (1991) established a resiliency/self-mulching index based on clay content, tendency to slake after puddling, and resistance to dispersion of the slaked material. An index value >15 indicates that the soil has a high resiliency (i.e., high tendency to self-mulch), a value <5 indicates virtually no resiliency, and index values between 5 and 15 indicate weak to moderate resiliency.

6. Sieving of soil aggregates

Sieving air-dry soil to obtain dry aggregate distribution, often termed *dry sieving*, is an early and still-used method to assess a soil's susceptibility to wind erosion (e.g., White, 1993). The original rotary sieve technique used in dry sieving (Chepil, 1952) has since been modified to account for the breakdown of aggregates during the sieving process so that the original or "field" aggregate distribution may be estimated (White, 1993; Black et al., 1989). An advantage of the method is that the data obtained can supply simple indices of dry soil aggregate distribution, notably aggregate mean weight diameter, MWD (MWD = mean aggregate diameter of a selected aggregate size class multiplied by the relative weight of aggregates in that size class, summed over all size classes). The percentage of soil aggregates with >0.84 mm is a commonly used index of the wind erodibility of soil. The greater the percentage, the lower the susceptibility to wind erosion.

Aggregate *wet sieving*, which is the sieving of water-saturated aggregates under water, is the most commonly utilized technique to determine aggregate size distribution, aggregate resistance to breakup through slaking, and aggregate stability to water (e.g., Angers and Mehuys, 1993). Although well established as the preferred method for the physical description of soil aggregates, a lack of standardization has resulted in a proliferation of wet sieving procedures resulting from differences in sample collection and preparation, and wetting and sieving techniques (Imhof, 1988). In addition, the moisture content of the aggregates at sampling and during storage prior to analysis can strongly influence the measured aggregate stability. As with dry sieving, the MWD is the primary index used to characterize the size distribution of water stable aggregates, and selected aggregate sizes (or size classes) are usually considered. In contrast to dry sieving, however, a correction for the presence of non-aggregated soil particles (e.g., sand grains) is usually employed to remove an occasionally substantial "primary particle bias" in the aggregate stability results (Angers and Mehuys, 1993).

Notwithstanding the above concerns and limitations in wet sieving technology, it has yielded several aggregate stability indices that have been widely accepted and reported in the literature (Imhof, 1988). For example, wet sieving is often used to determine the weight of water stable macroaggregates (>250 μm), which has been found to be a highly sensitive indicator of the impact of crop management practices on soil structural quality (Tisdall, 1996). Wet sieving is also used to distinguish the ability of aggregates to resist breakup through slaking versus breakup through mechanical impact; the former test uses initially air-dry aggregates, whereas the latter uses pre-moistened aggregates (Dexter, 1988). Kemper and Rosenau (1986) proposed a now widely used wet sieving method to determine aggregate stability to mainly mechanical stresses: initially air-dry 1–2 mm aggregates are rehumidified, then placed on a single 250-μm wet sieve, and then shaken at a standard rate for a standard length of time. It is often advised, however, that a nest of different sized sieves (generally ranging from 5 to 250 μm) be used to compare the distribution of water stable aggregates among several aggregate size classes (Angers and Mehuys, 1993; Beare and Bruce, 1993).

VII. INTEGRATING CAPACITANCE AND STRENGTH ATTRIBUTES

As shown in Tables 2.1 to 2.6, many critical parameters are available to address the various capacitor and strength aspects of soil physical quality. These parameters tend to be restricted or specialized to individual (single) soil processes (e.g., water storage, soil penetrability, etc.), and thus they tend to reflect poorly the connections and synergistic interactions that occur among soil processes. For specifying critical soil parameters, it is desirable, but not often possible, to integrate a number of these synergistic interactions among the processes involving soil, plant growth, and weather. Two primary approaches would seem to offer the greatest probability for application. It is feasible to integrate a number of single-process parameters into higher level parameters that better reflect the nature of relevant interactive processes. An alternative approach is to use dynamic, process-based simulation models that can handle a high level of soil, crop, and atmospheric complexity and interactions. Selected examples arising out of the earlier sections are given here.

Critical state soil mechanics provides a framework for interpreting soil strength, deformation, structure, and permeability interactions (Kirby and Blunden, 1991). Structure both affects and is affected by strength and deformation. Similarly, soil structure affects permeability. Three probable changes to soil structure occur as described in Section V depending on the magnitude of the applied stresses relative to P_C. Under stresses that are $< P_C$, little change in structure or permeability occurs as the soil is strong enough to resist the stresses. Typically in tillage operations, the compressive stress is small, but the shear stress is large. The soil expands and the changes in structure occur in thin zones with relatively undeformed soil between. The permeability may decrease or increase depending on which has greater influence, the disruption of pore continuity in the thin sheared zones or the increased pore volume. When compaction occurs beneath tires, both the shear and compression stresses are large, and the soil compresses. Changes in structure occur throughout the affected soil, and the permeability decreases, with the associated decreases in capacity for exchange of water and gases. Critical state soil mechanics provides a method for anticipating the effect of soil deformation on structure and permeability but as yet does not include a means of specifying the magnitude of these effects.

A recently proposed high-level integrating parameter for soil physical quality is the "Nonlimiting" or "Least Limiting" Water Range (Letey, 1985; da Silva et al., 1994). Simply stated, the nonlimiting water range, NLWR, is the range of volumetric water contents in the root zone over which a crop is neither restricted in its ability to acquire water or air, nor inhibited in its ability to extend and proliferate its root system. The NLWR is defined by upper and lower water content limits. The upper limit, θ_U, is the lesser of the field capacity water content, FC (see Section III), or the water content corresponding to an air-filled porosity of 10%, i.e., $\varepsilon_a = 0.1$ (Section IV). The lower limit, θ_L, is the greater of the permanent wilting point water content, PWP (Section III), or the water content corresponding to a cone penetration resistence of 2 MPa (i.e., PR = 2 MPa) (Section V). When the root zone water content is substantially outside the NLWR for a substantial period of time during the growing season, crop yield is reduced. The NLWR thus integrates several single-process soil storage and strength parameters (i.e., FC, $\varepsilon_a = 0.1$, PWP, PR = 2 MPa) and it does so in a way that is relevant to crop production. Although the NLWR concept requires more development and testing, Topp et al. (1994) used it successfully to show how agricultural practices affect soil physical quality from the perspective of sustained crop production, and da Silva (1995) showed that crop development is negatively correlated with the frequency with which the root zone water content falls outside the NLWR during the growing season. The NLWR thus shows considerable promise as a high-level soil physical quality parameter, and additional high-level parameters that relate to other aspects of crop production and ecosystem health should be developed.

Process-based simulation models can show the importance and impact of temporal events that occur both during the growing season (e.g., precipitation events, temperature changes) and between growing seasons (e.g., snow accumulation, depth of frost penetration, multi-season droughts). A fairly recent approach to estimating soil aeration is to predict the soil oxygen concentration profile in the root zone using a model based on soil gas diffusion (Eq. 8, Section IV; see also, Glinski and Stepniewski, 1985; Cook, 1995). This technique requires soil respiration to be measured or estimated as a function of depth (Cook, 1995), water content (Linn and Doran, 1984; Orchard and Cook, 1983; Cook et al., 1985; West et al., 1988), and temperature (Howard and Howard, 1979; Yoneda, 1975; Abrosimova, 1979; Ross and Cairns, 1978; Lloyd and Taylor, 1994). A measure of soil porosity is also required. In this approach, the limiting O_2 concentration for adequate soil aeration has been set at 10 g m^{-3} (Meyer and Barrs, 1991) and zero g m^{-3} (Blackwell and Wells, 1983). Modelling the soil O_2 profile is potentially a good indicator of soil aeration, because it integrates the established parameters and mechanisms of soil gas diffusion and water content. It also accounts indirectly for soil gas storage because D_g (gas diffusion coefficient) depends on both ε and ε_a (e.g., Millington and Quirk, 1961). A possible limitation is that the approach neglects soil gas advection.

As mentioned in Section III, process-based models of soil water, budget, and crop growth are being used increasingly to develop land and crop management recommendations, such as whether to fallow or plant crop, what type and rate of fertilization to use, what type of crop to plant based on the amount of water stored in the root zone at seeding time, probable (predicted) crop yield, and what levels of crop insurance compensation are appropriate after an adverse growing season (e.g., Akinremi and McGinn, 1996; Ash et al., 1992; de Jong and Bootsma, 1996; O'Brien, 1992; Raddatz, 1992; Raddatz et al., 1996). Although process-based models admittedly have the disadvantage of requiring considerable (and accurate) input data, continuing development of data bases and pedotransfer function technology (e.g., Kay et al., 1997; Wösten, this volume) greatly increase the feasibility of using such models to manage the soil for optimum crop production and ecosystem health.

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Chapter 3

CHEMICAL ATTRIBUTES AND PROCESSES AFFECTING SOIL QUALITY

D. HEIL and G. SPOSITO

I. INTRODUCTION

The primary function of soil in relation to its chemical quality for crop production is to provide nutrients for crop growth. Because of the use of pesticides and fertilizers and the application of sewage sludges and other wastes to agricultural lands, the capacity of a soil to immobilize pesticides and detoxify and immobilize heavy metals must also be considered when determining the chemical aspects of soil quality. Three chemical aspects of soil quality are discussed in this chapter: nutrient availability, phytotoxicity of trace metals, and pesticide mobility. Soil chemical properties that affect one or more of these three aspects are cation exchange capacity (CEC), pE, pH, electrical conductivity (EC), exchangeable sodium percentage (ESP), organic matter content, and mineralogy. These properties, or chemical attributes of soil quality, may be used to assess soil quality and to monitor changes caused by degradation (Larson and Pierce, 1991; Arshad and Coen, 1992; Karlen and Stott, 1994).

The soluble concentration of an element or compound often determines plant nutrient uptake, phytotoxicity, and contaminant mobility in soils. The dependence of these soil aspects on soil chemical properties (attributes) can be explained based on the chemical speciation of a nutrient or contaminant in the soil. Chemical speciation is the distribution of an element or compound among all of the chemical forms present in solid, aqueous, and gaseous phases, and is controlled by chemical processes (reactions) occurring in the soil environment. A knowledge of the chemical reactions controlling the speciation of an element or compound in a soil assists in predicting how a change in one or more of the soil chemical attributes will affect nutrient availability and also contaminant biotoxicity and mobility. Therefore, following a description of the soil chemical attributes, we describe soil chemical reactions and then discuss processes through which the soil chemical attributes affect the three soil aspects.

II. SOIL CHEMICAL ATTRIBUTES

A. Mineralogy

Soluble mineral forms of plant nutrients can be supplied via mineral dissolution. The mineral content of a soil is a function of the mineralogy of the parent material from which the soil formed and soil-weathering factors. When a mineral form of an element is not present in the soil or when the kinetics of dissolution or precipitation of a mineral form of the element are slow, the partitioning of that element is dominated by adsorption, especially on a short time scale. The soil mineral colloids present in a soil determine the types of inorganic mineral functional groups that are present, and consequently determine the nature of adsorption processes that occur. Measured soil properties that have proven useful in predicting adsorption and the effects of adsorption on the bioavailability and mobility of elements in soils include the content and type of layer silicate clay minerals and the content of Fe and Al oxide and hydrous oxide minerals. Analytical methods are given for the quantification of clay minerals and Al and Fe oxide and hydrous oxide minerals (Klute, 1986).

B. Organic matter

Soil organic matter is a source of plant nutrients, which may be released into a plant available form via decomposition by microorganisms. Humus is highly decomposed organic matter containing reactive adsorption sites or "functional groups" that retain cations in the soil system. Base nutrient cations (Ca, Mg, K) and ammonium are retained as exchangeable cations. Trace element cations are retained more strongly due to specific chemical adsorption. The cation exchange capacity of organic matter is high even compared to the 2:1 layer silicate minerals, and thus organic matter contributes substantially to the cation exchange capacity of soils. Organic matter is a dominant factor in the removal of pesticides from soil solution. Procedures for the measurement of soil organic matter content are given by Page et al. (1982).

C. pE and pH

The pE and pH of soil solutions have a profound effect on element solubility in soils. Oxidation/reduction or "redox" reactions control the solubility and speciation of elements that can exist in multiple oxidation states under the range of conditions existing in soils, especially for N, S, Fe, Mn, and Se. The redox status of a soil may be evaluated as the availability of electrons that may participate in redox reactions, which may be expressed by the negative logarithm of the free electron activity (Sposito, 1989):

$$pE = -\log(e^-) \qquad [(\ ) = \text{activity}] \tag{1}$$

The pE value of soils is controlled by physical soil conditions (water content and porosity) and biological activity in the soil. Values of pE in soil range from −6.0 to 13.0. Methods for measurement of soil redox status are given by Bartlett and James (1993).

The processes of mineral dissolution and also adsorption at acidic functional groups are dependent on pH, in addition to the fact that cation exchange capacity depends on pH. Acidic soils are often associated with nutrient deficiencies of the base nutrient cations Ca, Mg, and K, and also deficiencies of P. Trace element cations and Al are often present at high enough concentrations to cause toxicity to plants in acidic soils. Alkaline soils are associated with deficiencies of trace element nutrients, in particular Fe and Zn, and also deficiencies of P. The effect of pH on the availability of a given nutrient can often be explained by either a precipitation or an adsorption model. For example, the decrease in P availability in acidic conditions may be explained in terms of precipitation of the Al and Fe phosphate minerals variscite and strengite (Lindsay, 1979), or adsorption of phosphate by metal oxide functional groups (Goldberg, 1992).

D. Electrical conductivity and exchangeable sodium percentage

Salinity limits crop production when the concentration of soluble salts in the soil solution is high enough to decrease absorption of water. An electrical conductivity measurement of a saturated paste extract provides an estimate of the total soluble salts (Page et al., 1982). The critical electrical conductivity at which growth is affected depends on plant species (Bohn et al., 1985). In arid and semi-arid regions, the presence of exchangeable sodium can cause swelling of soil aggregates and dispersion of clay, which results in a decrease in hydraulic conductivity (Shainberg and Letey, 1984).

E. Cation exchange capacity

Exchangeable cations are cations that are adsorbed weakly by soil particles and can be easily displaced from the soil particle surface into the solution phase by another cation. Cation exchange capacity is measured as the total number of equivalents of cations displaced per unit mass of soil solids by an extracting solution containing a high concentration (usually 1.0 M) of an extracting cation. Potassium or ammonium are often used as the extracting cation. Detailed methods for the determination of cation exchange capacity are given by Page et al. (1982).

The cation exchange capacity of organic matter, 1:1 clay minerals, and Al and Fe oxide minerals is highly dependent upon pH. In the case of the 1:1 mineral kaolinite and metal oxide minerals, the particle surface becomes positive at acidic pH and these colloids lose their capacity to retain exchangeable cations. Consequently, soil organic matter plays an extremely important role in the retention of nutrient cations in highly weathered acidic soils, in which 2:1 clay minerals possessing permanent charge are not present (Coleman et al., 1989).

It is important to recognize that an individual cation may not be retained in substantial amounts at cation exchange sites, even if the soil cation exchange capacity is high. In order for a cation to occupy exchange sites, it must compete favorably with other cations present in the soil solution. For cations that react in a specific manner with soil functional groups and form inner-sphere complexes, the cation exchange capacity does not necessarily provide a good estimate of the amount of cation that may be adsorbed from solution. For example, Fe-oxide minerals adsorb a significant amount of metals at pH values at which the cation exchange capacity of the Fe-oxide mineral is expected to be very small (McKenzie, 1980).

III. SOIL CHEMICAL PROCESSES

The chemical processes through which the designated chemical attributes affect crop production include the formation of both organic and inorganic soluble complexes, precipitation/dissolution of solid minerals, adsorption, and oxidation/reduction. As previously discussed, both the bioavailability and mobility of an element in a soil depend largely upon the partitioning of the element between soil solids and the soil solution. The solubility of an element in a soil at a given time is affected by several reactions occurring simultaneously. Batch sorption experiments yield information on element partitioning, which can be used to model nutrient bioavailability (Barber, 1995). However, such experiments do not allow for the identification of the chemical reactions controlling element partitioning. The element partitioning in soils is often found to change with time, indicating that the reactions involved are rate limited. The relative rate of reactions occurring in soils is: soluble complex formation > adsorption > mineral dissolution (Sparks and Suarez, 1991).

A. Soluble complexes

The formation of soluble complexes may be modeled using thermodynamic constants if the total soluble concentrations or activities of all cations and anions present in the soil solution are quantified. The major inorganic cations that are present in soil solutions include Ca^{2+}, Mg^{2+}, K^+, Na^+, and NH_4^+, with the soluble concentrations of Fe^{3+} and Al^{3+} (and H^+) significant in acid conditions. Major inorganic anions in soil solutions include Cl^-, F^-, NO_3^-, HPO_4^{2-}, $H_2PO_4^-$, SO_4^{2-}, with significant soluble concentrations of HCO_3^- (and OH^-) in alkaline conditions.

Soluble organic complexes may form in the presence of either naturally occurring or synthetic soluble organic compounds. Important naturally occurring soluble organic compounds are simple aliphatic acids, amino acids, sugar acids including monosaccharides and polysaccharides, and phenolic compounds (Stevenson, 1994). Typical concentrations of these organic compounds in soil solutions are reported by Sposito (1989). Soluble humic and fulvic acids may also be present in the soil solution at significant concentrations.

The formation of soluble complexes of an element often results in an increase in the total soluble concentration of that element in a soil solution. The transport of an element in soils by both convection (mass flow) and diffusion increases with the soluble concentration of that element (Nelson et al., 1983). It follows that the mobility of an element in the soil will be increased as a result of soluble complex formation. However, the effect of complex formation on the bioavailability of an element depends on the ability of the plant root to absorb the complexed species or break down the complex into the free species near the root surface, so that absorption of the free ion form of the element may take place (Marschner, 1986). Experimental studies on the effects of complex formation on plant uptake of nutrients are reviewed by Ritchie and Sposito (1995).

B. Sorption

A sorption isotherm is a plot of the amount of an element retained in the solid phase (q) as a function of the soluble concentration of that element (c_1). The term sorption does not imply any specific reaction mechanism, only that a portion of the element is associated with a solid phase. Sorption isotherms may be characterized by four different types (Fig. 3.1). The shape of a sorption isotherm may be interpreted on the basis of the relative affinity of the sorbed species for the solution and solid phases (Sposito, 1984). Sorption isotherm data may be fitted to the Langmuir or Freundlich equations (Sposito, 1989). Both of these equations may be used to

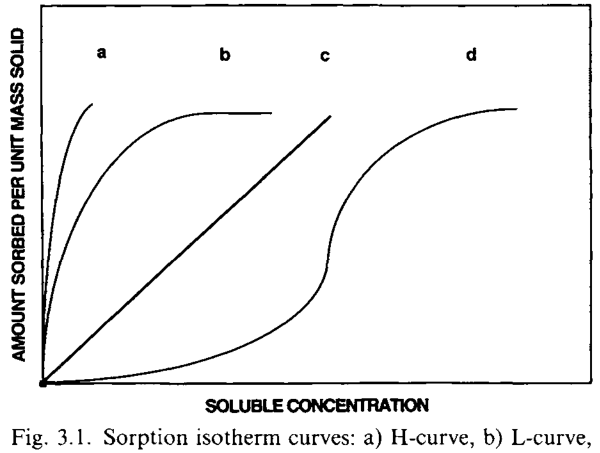

Fig. 3.1. Sorption isotherm curves: a) H-curve, b) L-curve, c) C-curve, d) S-curve.

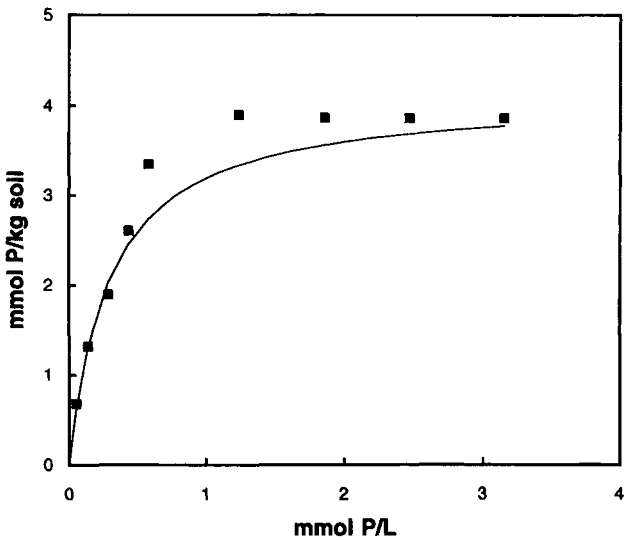

Fig. 3.2. Phosphate sorption isotherm for Colby series soil.

calculate the amount sorbed as a function of the soluble concentration. Figure 3.2 is an example of a phosphate sorption isotherm for a soil, with the isotherm curve drawn using a best fit of data to the Langmuir model.

When two cations are competitively sorbed, a cation exchange model may be used to express the partitioning of the two cations. For exchange of Ca for K, the reaction may be written as:

$$\mathrm{K}X + 1/2\mathrm{Ca}^{2+} = \mathrm{K}^+ + \mathrm{Ca}_{1/2}X \tag{2}$$

where X designates one mole of negative charge on the exchanger solid.
The Gapon equation for this reaction may be expressed as:

$$\frac{q_{\mathrm{K}}}{2q_{\mathrm{Ca}}} = \mathrm{K_G}\frac{[\mathrm{K}^+]}{[\mathrm{Ca}^{2+}]^{1/2}}\frac{\gamma_{\mathrm{K}}}{\gamma_{\mathrm{Ca}}^{1/2}} \tag{3}$$

where q_{K} and q_{Ca} are the amounts of exchangeable K and Ca in units of moles per kg of soil, and γ_{K} and γ_{Ca} are activity coefficients for K^+ and Ca^{2+} in the solution phase. Cation exchange models may be used to describe sorption isotherms for exchangeable cations under certain conditions. For example, the Gapon equation has been used to model sorption isotherms for K in soils by applying the above equation with the assumption that the soluble concentration of Ca (and also Mg) remains nearly constant (Barber, 1995).

C. Precipitation/dissolution

Precipitation is defined as the accumulation of substances to form a new solid phase. A solid form of an element added to a soil as a fertilizer or contaminant may precipitate if a sufficient concentration remains after adsorption to supersaturate the solution with respect to a certain solid phase.

Mass-action expressions may also be used to predict the solubility of an element in equilibrium with a solid phase. For dissolution of the mineral hydroxyapatite (Lindsay, 1979)

$$Ca_5(PO_4)_3OH + 7H^+ = 5Ca^{2+} + 3H_2PO_4^- + H_2O \tag{4}$$

the corresponding equilibrium constant has a value of log $K^o = 14.46$. The solubility of P in equilibrium with hydroxyapatite may be expressed as:

$$\log(H_2PO_4^-) = 14.46 + 5/3 \log(Ca^{2+}) - 7/3\,pH \tag{5}$$

This equation may be used to calculate P solubility between pH 2.2 and 7.2, because $H_2PO_4^-$ is the dominant form of soluble P in this pH range. Equation 5 shows that P solubility in equilibrium with hydroxyapatite depends upon both pH and Ca^{2+} activity. At a fixed (Ca^{2+}), the activity of the $H_2PO_4^-$ species increases as pH decreases. This relationship is consistent with observations that rock-phosphate (hydroxyapatite) is not an effective fertilizer source of P in alkaline soils (Troeh and Thompson, 1993).

D. Adsorption

Adsorption is defined as the accumulation of a species at the surface of an existing solid. Chemical species may be adsorbed at the solid–solution interface by three mechanisms, identified as the inner-sphere surface complex, the outer-sphere surface complex, and the diffuse ion swarm (Sposito, 1989). The reactive sites on particle and/or molecular surfaces where adsorption occurs are termed "functional groups". Functional groups on soil colloids include metal hydroxy groups located at the surface of metal oxide and hydrous oxide minerals and also at the edges of layer silicate clay minerals, the siloxane cavity on the basal plane of 2:1 layer silicate clay minerals, and acidic organic hydroxyl groups on humus (ibid.).

Numerous investigations have shown that the metal oxide functional groups existing at the surface of Fe, Al, and Mn oxide minerals and the edges of clay minerals adsorb both cation and anion species from solution. In particular, trace metal cations, such as Cu, Pb, Zn, and Cd, and oxyanions, such as phosphate, chromate, and selenite, may be adsorbed tenaciously by metal oxide minerals as a consequence of inner-sphere complex formation. Adsorption of both cations and anions is highly pH-dependent, with the adsorption of cations enhanced at high values of pH (McKenzie, 1980) and the adsorption of anions enhanced at low pH (Goldberg and Sposito, 1984). This mechanism causes fixation of P in highly weathered, acidic soils. Wendt et al. (1993) found that adsorption of P by Niger soils increased with iron oxide content, as measured by dithionite-citrate-bicarbonate Fe and oxalate-extractable Fe. Increased adsorption of P in soils with high levels of Fe oxides caused a decrease in the soluble concentration of P and, consequently, a decrease in the yield of millet grain in response to P fertilizer additions.

The siloxane cavity corresponds to a hexagonal ring of oxygen atoms located on the basal plane of layer silicate clay minerals. The reactivity of the siloxane cavity results from the negative charge surplus originating from isomorphic substitution of cations within the clay structure, often referred to as permanent charge. For clay minerals such as mica, illite, and vermiculite, where a significant extent of substitution occurs in the tetrahedral sheet, the negative electrical charge is located directly adjacent to the siloxane surface. This localization of charge is responsible for the formation of inner-sphere surface complexes with K and ammonium by the siloxane cavity functional groups on these clay minerals (Sposito, 1984). Potassium in this form is not extracted by cation exchange procedures, and is termed non-exchangeable or fixed. Soils in the San Joaquin Valley, Calif., U.S.A., that contained vermiculite and mica in the clay fraction required "enormous" additions of K fertilizer to overcome K deficiencies in cotton compared to soils in other locations (Cassman et al., 1992). Long-term crop production without K fertilizers resulted in depletion of plant available (exchangeable) K in these soils, corresponding to a shift to the left on an adsorption isotherm curve (such as those depicted in Fig. 3.1).

Carboxylic and phenolic groups are present in relatively large quantities in soil humus and play an important role in the binding of metal cations (Stevenson, 1994). Adsorption at these sites results in immobilization of potentially toxic metals and also retention and buffering of micronutrients. The strength of binding of metal cations by soil humic acid follows the order Cu > Pb > Cd > Zn (ibid.). The high affinity of organic functional groups for Cu often results in a high proportion of the total Cu in soils being associated with the organic matter fraction. A large proportion of the Cu in soils that equilibrates with the soil solution is complexed to organic matter, so that organically bound Cu is an important source of this micronutrient in soils (Barber, 1995). Plant deficiencies of Cu have been observed in soils with very high organic matter contents, although these deficiencies are attributed to low total Cu concentrations in organic soils (Marschner, 1986). The kinetics of adsorption and release of adsorbed metal cations by humus are rapid, with the time required to reach equilibrium on the order of a few minutes (Sparks, 1989). The overall effect of adsorption of cations by humic substances on the partitioning of the cation between solid and solution phases is dependent on the solubility of humic and fulvic acids in the soil solution. The concentration of metal cations in a soil solution may be significantly increased in the presence of a high soluble concentration of humic and fulvic acids (MacCarthy et al., 1990).

Sorption of plant nutrients can either increase or decrease plant nutrient uptake from soils. Numerous studies indicate that sorption of a nutrient leads to a decrease in plant uptake if the soluble concentration is decreased as a result of sorption (Barber, 1995). Theoretically, sorbed species are completely released into solution if soluble concentration is maintained at zero. However, since a minimum soluble concentration of a nutrient is required to cause net nutrient uptake (ibid.), plant growth is severely stressed before this condition is met. Nutrient solution experiments provide information on the critical soluble nutrient concentrations for plant growth. When a high amount of a nutrient is adsorbed at a very low soluble concentration of the nutrient, as occurs with an H-curve isotherm, the soluble concentration of the nutrient exceeds the critical level for plant growth only when the total concentration of the nutrient in the soil is very high. The strength of binding is related to the mechanism of adsorption, and inner-sphere complexation often results in an H-type isotherm. Subsequently, inner-sphere complex formation tends to result in the fixation of nutrients (Sposito, 1989). Examples include P adsorption by Al and Fe oxides, and K adsorption by illite and vermiculite. These nutrients will become available to plants only when the total concentration of the element in the soil is high enough to shift the position on the adsorption isotherm to a point corresponding to a higher soluble concentration. In general, retention of nutrients in a plant-available form is favored by outer-sphere and diffuse-swarm mechanisms of adsorption, by which nutrients may be released into solution in the soluble concentration range in which plant growth is optimal. Conversely, the immobilization of pollutant chemicals is favored by inner-sphere complexation, in which a high amount of adsorption coincides with a very low soluble concentration.

E. Oxidation/reduction

Oxidation/reduction or "redox" reactions involve the transfer of electrons from one chemical species to another. An oxidation/reduction reaction produces a change in the oxidation state of the species involved in this electron transfer. A complete redox reaction always designates a species that provides an electron and another species that accepts the electron. The electron source is often a biologically mediated reaction, such as organic matter oxidation. The reduction of inorganic elements in soils may be expressed with a half-reaction, in which the source of the electron is not explicitly shown. For example, the reductive dissolution of an Fe(III) hydroxide mineral may be expressed as the half-reaction:

$$Fe(OH)_{3(s)} + 3H^+ + e^- = Fe^{2+} + 3H_2O \qquad (6)$$

The designation of the free electron activity of a soil solution allows the application of a thermodynamic equilibrium equation to redox reactions. The corresponding equilibrium constant for the above reaction is $\log K^o = 16.4$. The solubility of the Fe hydroxide mineral can be calculated with the equation:

$$\log(Fe^{2+}) = 16.4 - 3pH - pE \qquad (7)$$

The activity of the Fe^{2+} species increases with increased free electron activity. The solubility of both Fe and Mn in soils increases dramatically under reducing conditions (low values of pE).

Reduction half-reactions may also be used to determine the dominant soluble form of an element. The reduction of selenate (Se VI) to selenite (Se IV) is illustrated by the following half-reaction (Sposito et al., 1991):

$$1/2SeO_4^{2-} + 3/2H^+ + e^- = 1/2HSeO_3^- + 1/2H_2O \qquad (8)$$

with a corresponding equilibrium constant of $\log K^o = 18.2$. The ratio of the activities of the selenite and selenate species may be expressed as:

$$\log \frac{(HSeO_3^-)}{(SeO_4^{2-})} = 18.2 - 1.5\,pH - pE \tag{9}$$

This equation can be used to calculate the predominant soluble species of Se if the pH and pE are known. For example, at a pH of 7.0, the activities of the selenate and selenite species will be equal at pE = 7.7. At values of pE < 7.7, selenite will be the dominant species, and at pE values >7.7, selenate will be the dominant species. These relationships are paramount to the mobility of Se in soils because selenite is adsorbed strongly in soils, whereas selenate is not.

IV. BIOAVAILABILITY OF PLANT NUTRIENTS

From the perspective of soil chemistry, the bioavailability of an element is largely determined by competition among the plant root system, the soil solution, and solid phases (Fraústo da Silva and Williams, 1991; Sposito, 1989). This point of view is illustrated in Figure 3.3 (Sposito, 1989) for any element that can exist as an ionic solute in the soil solution (e.g., Ca or Al, S or Se).

Besides an abundant free ion (or free molecule) species population in the soil solution, effective competition for an element by plant roots requires the element to move to a plant root surface on a time scale that is relevant to plant growth and development (Barber, 1995). Within the rhizosphere, the movement of nutrient elements (as well as toxic elements) involves diffusion and convective flow in water over distances in the range 0.1–15 mm, depending on the root population density in soil. Convective flow of an element occurs only when a plant transpires and its roots

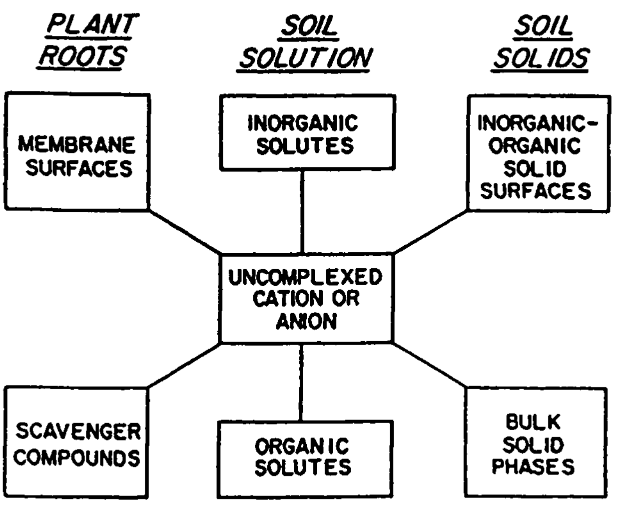

Fig. 3.3. Schema of the competition among complexing ligands associated with plant roots, the soil solution, and soil solid phases for "free" aqueous ions. Reprinted with permission from G. Sposito, The chemistry of soils, Oxford University Press, New York, 1989.

absorb water from the rhizosphere. The rate of water absorption (volume per unit root surface area per unit time) is typically on the order of μm s^{-1} (approximately the same as dm day^{-1}). Thus, for a soil solution concentration of the order of M m^{-3}, an element uptake flux on the order of μM m^{-2} s^{-1} is expected. (*Flux* is the amount absorbed per unit cross-sectional area of root surface per unit time, equal numerically to the product of concentration and the water absorption rate.) Taking mm as a representative distance scale over which convective flow to the root occurs, one estimates a corresponding time scale of hours for convective nutrient uptake. This time scale is consistent with the fact that water absorption takes place during daylight only. For relatively mobile species, such as NO_3^-, $H_3BO_3^0$, and Ca^{2+}, uptake by convective flow can be a dominant mechanism in fertile soils (ibid.).

These considerations of competition and transport with the rhizosphere lead to a definition of bioavailability in soil chemical terms (Sposito, 1989): *An element is bioavailable if it is present as, or can be transformed readily to, the free ion species; if it can move to plant roots on a time scale that is relevant to plant growth and development; and if, once absorbed by the root, it affects the life cycle of the plant.* The adverb "readily" means "on a time scale relevant to the continued growth of a plant." The phrase "affects the life cycle of the plant" means "produces growth and development" in the case of an essential element, whereas for a toxic element it means "produces phytotoxicity".

A. Soil speciation and bioavailability

The existence of an element in different chemical forms in gaseous, solid, or aqueous solution phases provides a conceptual basis for speciation in soils. A "chemical species" in soil refers either to a specific molecular arrangement of the atoms of an element or, more simply, to the result of an operational process of detection and quantitation aimed at elucidating chemical form (Bernhard et al., 1986). In principle, the former definition should be the outcome of the latter methodological definition; in practice, this close connection is difficult to achieve in natural systems. Speciation can be described further by considering the type of bonding between an element and the constituents of soil. For example, an element in free ionic form may be bound to clay minerals or to organic matter merely by coulombic forces, whereas it may form covalent bonds with surface functional groups on hydrous oxides. In solution or on an adsorbent, ligands can form inner- or outer-sphere complexes with cations. The speciation of an element in soil is always transitory, because soil is only one "compartment" of the biogeochemical cycles of elements in an ecosystem. The residence time of an element can vary considerably depending on the mobility of its predominant chemical species and the rate of evolution of soil. For example, Al, Fe, and Si are the most abundant elements in soils (Sposito, 1989), but usually $<1\%$ of their soil chemical species cycle in one year, because most of their chemical forms are extremely immobile (Sposito and Page, 1984). This slow cycling is a positive attribute in respect to the toxicity of Al to plants, but is negative in respect to Fe nutrition. Thus, slow cycling becomes a problem if it can lead to lessened bioavailability of elements required for plant growth (e.g., Fe, Cu, and Zn).

B. Nutrient supply

Mobile species, such as NO_3^-, are taken up by plants on a time scale of hours (mm distance scale divided by a water absorption rate on the order of μm s^{-1}). If this convective uptake time scale is much larger than the time scale for water to percolate through the soil profile, a mobile species is leached instead of taken up by plants. Percolation time scales through the rooting zone can range broadly, from tens of minutes to days, depending on soil texture, structure, and water content (Hillel, 1980). Thus, leaching losses are expected for mobile species in permeable soils.

Diffusive uptake is usually a much slower process than convective uptake, but it can be very important for relatively immobile elements, such as P and Cu, especially when their total soil solution concentrations, c_l, are low (Barber, 1995). The effective diffusion coefficient of an element is proportional to θ, the volumetric water content, and inversely related to the soil *buffer power*, β_e, (Nye, 1979; Barber, 1995)

$$\beta_e = \frac{dc_s}{dc_l} \tag{10}$$

where c_s is the bulk soil concentration (M m^{-3} soil).

The interplay of soil solution concentration, buffer power, and diffusion time scale can be illustrated by the case of P uptake by corn seedlings grown in pot culture on an Alfisol and two Mollisols (Schenk and Barber, 1979). Values of the soil parameters c_l, β_e/θ, and τ_D, the diffusion time scale (Sposito, 1989), are listed in Table 3.1. The Alfisol had twice the soil solution P concentration but half the buffer power and diffusion time scale as one of the Mollisols, whereas it had one-third the soil solution concentration and seven times the buffer power and diffusion time scale of the other Mollisol. Despite these differences, the P uptake by corn grown on the

TABLE 3.1

Values of soil uptake factors and P uptake by corn grown for seven days on three soils[a]

Soil	c_l (mmol m^{-3})	β_e/θ	P supply[b] (mmol m^{-3})	τ_D[c] (day)	τ_{Def}[d] (day)	P uptake[e] (μmol pot^{-1})
Ochraqualf	45	21	945	137	6.5	540
Argiaqoull	20	52	1040	336	6.5	401
Argiudoll	126	3	378	23	7.7	393

[a]Data from Schenk and Barber (1979).
[b]Product of c_l and (β_e/θ) (Barber, 1995).
[c]$\tau_D = 2\delta^2/D_e$, δ = distance diffused $\cong$ 3 mm, D_e = effective diffusion coefficient (Sposito, 1989).
[d]Ratio of τ_D to (β_e/θ).
[e]Values not significantly different (P = 0.05) according to Duncan's multiple-range test.

three soils (last column in Table 3.1) was the same, within experimental variability (Schenk and Barber, 1979).

This result can be understood once the typical inverse dependence of D_e on β_e is considered. An "effective diffusion time scale", τ_{Def}, can be defined as the ratio of τ_D and β_e/θ, to compensate for the expected proportionality between τ_D and β_e (since τ_D is inversely proportional to D_e; see Sposito, 1989). The values of τ_{Def} for the three soils (sixth column in Table 3.1) are essentially equal. Thus, if sufficient P can be supplied by the soils, the movement of P into the plant should be about the same, as measured by τ_{Def}. The P-supplying capability of a soil can be estimated as the product of c_l and β_e/θ (ibid.). It is shown in the third column of Table 3.1. The Alfisol and one of the Mollisols have about three times the P supply as the other Mollisol, but all three values are actually quite large relative to unfertilized soils (ibid.) and, therefore, were sufficient to supply P needed by corn during the 17-day growing period.

This example points to the key role played by the buffer power in determining both the P supply and the effective diffusion time scale. It is this parameter that gives the clearest indication of soil quality in respect to nutrient bioavailability. Large values of β_e/θ lead to high P supply and low effective time scales for diffusion. Large values of the buffer power reflect large slopes in soil desorption isotherms (i.e., large releases of an adsorbed nutrient when its soil solution concentration is reduced by plant uptake or leaching). The key role, then, to soil quality with respect to nutrient uptake is labile adsorption processes.

V. PHYTOTOXICITY OF TRACE METALS

Some metals are essential to plant growth, whereas others are hazardous to plant growth (Troeh and Thompson, 1993). The essential metals (Mg, K, Ca, Mn, Fe, Cu, Zn, Mo) have the distinguishing property that they are among the "light elements" in the Periodic Table and have relatively small atomic radii (Sposito, 1986). Optimal soil solution concentrations of the essential metals for the nutrition of plants have not been found because of interdependencies among the biochemical functions of these elements (Fraústo da Silva and Williams, 1991). At a sufficiently high concentration of any element, however, deleterious effects on plant growth can be expected. This conclusion applies particularly to the micronutrient metals (Mn, Fe, Cu, Zn, Mo), since the need for them is small to begin with. At some soil solution concentration, which is plant species-dependent, each of these metals can produce phytotoxic effects (ibid.; Troeh and Thompson, 1993).

The greater concern, however, is for those metal elements in the Periodic Table whose abundance in the hydrosphere or atmosphere has been increased significantly by the accelerated extraction of minerals and fossil fuels from the lithosphere, and by waste output from other technological processes characteristic of the present century. Most of these metal elements are typically at total concentrations well below 1 mM m^{-3} in pristine fresh waters. For this reason they are referred to as trace metals (Adriano, 1986). Rapid, localized increases of trace metal concentrations in the hydrosphere or atmosphere are observed concomitantly with the development of exploitative technologies. This kind of sudden change exposes the local biosphere, including the soil on which it depends, to ecological destabilization, thereby generating an institutional classification of the metals as "potentially hazardous" (Bernhard et al., 1986; Adriano, 1986). The extent to which a trace metal actually poses an environmental hazard depends not only on the rate and magnitude of its enrichment in the hydrosphere, atmosphere, or soil, but also on its chemical speciation as discussed in Section IV. A., and on the details of its biogeochemical cycling (ibid.).

A list of potentially hazardous trace metals is provided in Table 3.2 (Sposito, 1986). The list is not exhaustive, even to the extent of representing all of the environmentally important oxidation states of any one metal, but it is likely to include most of the metals slated for regulatory action on the basis of potential impact on the biosphere impact (Adriano, 1986). Besides the metals themselves, three important chemical parameters related to atomic structure also are listed in Table 3.2. The third column gives the ionic potential (IP) (Sposito, 1989):

$$\mathrm{IP} = \frac{\text{oxidation number of free cation}}{\text{radius of unsolvated cation}} \tag{11}$$

TABLE 3.2

Atomic structural properties of some potentially hazardous trace metals (Sposito, 1986)

Trace metal (Z)	Ionic radius (R)[a] (nm)	Ionic potential (IP)[b] (nm^{-1})	Misono parameter (Y) (nm)
Li(I)	0.074	13.5	0.053
Be(II)	0.027	74.1	0.032
Al(III)	0.053	56.6	0.073
V(IV)	0.059	67.8	0.211
Cr(III)	0.062	48.4	0.226
Mn(II)	0.083	24.1	0.273
Fe(II)	0.078	25.6	0.291
Co(II)	0.075	26.7	0.270
Ni(II)	0.069	29.0	0.252
Cu(II)	0.073	27.4	0.284
Zn(II)	0.075	26.7	0.240
Sr(II)	0.113	17.7	0.202
Mo(VI)	0.042	142.9	0.092
Ag(I)	0.115	8.7	0.405
Cd(II)	0.095	21.1	0.303
Sn(IV)	0.069	58.0	0.194
Sb(III)	0.077	39.0	0.254
Cs(I)	0.170	5.9	0.264
Hg(II)	0.102	19.6	0.396
Pb(II)	0.118	16.9	0.393

[a]From data compiled by Shannon and Prewitt (1969, 1970).
[b]For a discussion see Sposito (1989).

This parameter is used to classify metals according to the typical chemical species they form by interaction with nearest-neighbour water molecules in dilute aqueous solution at equilibrium with atmospheric CO_2 (pH 5.6). If (in units of nm^{-1}) $Z/R < 30$, a metal cation will be only solvated by interaction with water molecules; if $95 > Z/R > 30$, a metal cation can repel protons strongly enough from a solvating water molecule to form a hydrolytic chemical species; If $Z/R > 95$, the repulsion is strong enough to allow the formation of oxyion species (Carroll, 1970). Typical examples of trace metals in the three categories are: Ag (8.7), Al (56.6), and Mo (142.9), with their ionic potentials given in parentheses.

The biological significance of IP can be appreciated readily by reference to Banin-Navrot plots (Banin and Navrot, 1975), a log-log graph of the biological enrichment factor (EF_B),

$$EF_B = \frac{\text{element concentration in organism}}{\text{element concentration in crustal rock}} \tag{12}$$

plotted against the ionic potential. Two Banin-Navrot plots are shown in Figure 3.4. They refer to terrestrial plants and animals. Their shapes are remarkably similar and are very much like those of Banin-Navrot plots for soil microflora (bacteria and fungi) (ibid.). In general, EF_B displays a shallow minimum for IP in the range 11–15 nm^{-1} (Li^+, Ba^{2+}); a maximum near IP $\approx$ 22 nm^{-1}; then a precipitous drop to a deep minimum at IP in the range 45–60 nm^{-1}; and finally, a gradual rise toward very large values of IP > 100 nm^{-1}. The sharp drop in EF_B, by almost three orders of magnitude, occurs at IP $\approx$ p; 30 nm^{-1}, and the marked rise in EF_B after this drop occurs at IP $\approx$ 95 nm^{-1}. Therefore, one infers from Figure 3.4 that EF_B increases in the order hydrolyzed species « solvated species < oxyanion species. This order reflects the concept that essential elements are those that tend not to form hydrolyzed

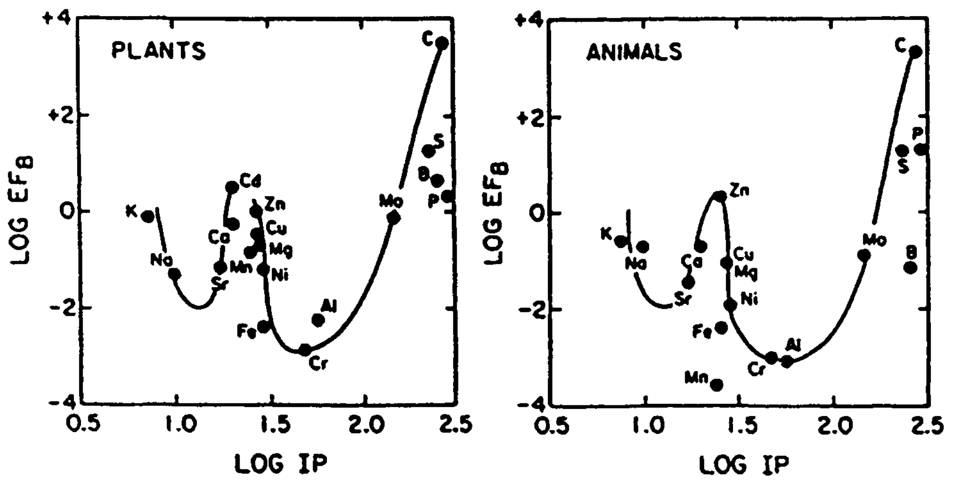

Fig. 3.4. Banin-Navrot plots for terrestrial plants and animals. Reprinted with permission from G. Sposito, The Chemistry of Soils, Oxford University Press, New York, 1989.

species that could adsorb or precipitate from aqueous solution. The micronutrient metals are perched along the "hydrolysis cliff" in a Banin-Navrot plot, whereas Al and Cr are at the bottom of the "insolubility chasm". (The shallow minimum that separates K from Ca may be associated with the relatively low crustal abundance of metals such as Li, Ba, and Sr.) These trends support the notion that solubility is a determining factor in essentiality versus phytotoxicity. If plants have evolved under conditions in which a metal is at low soil solution concentrations, they will not have developed biochemical pathways by which to detoxify the metal when it is at high soil solution concentrations.

The fourth column in Table 3.2 lists values of the Misono softness parameter (Y) (Misono et al., 1967; Adriano, 1986). The Misono parameter provides a quantitative measure of the tendency of a metal cation to be polarizable and enter into covalent bonding. The values of Y can be used to give quantitative significance to the classification of metal cations as "hard", "borderline", or "soft" Lewis acids (Fraústo da Silva and Williams, 1991). "Hard" Lewis acids are molecular units characterized by $Y < 0.25$ nm, high electronegativity, and low polarizability (e.g., Li or Al). They tend to form complexes and primarily coulombic interactions. "Soft" Lewis acids, on the other hand, are characterized by $Y > 0.32$ nm, low charge density, low electronegativity, and high polarizability (e.g., Hg or Pb(II)). They tend to form strong complexes stabilized by large enthalpy decreases and covalent interactions. Values of Y between 0.25 and 0.32 nm correspond to "borderline" Lewis acids whose tendency to covalency depends on whether specific solvent, stereochemical, and electronic configurational factors are present. The "borderline" Lewis acids include the micronutrient metals, Mn(II), Fe(II), Cu, and Zn. Note that metal cations whose IP values fall into the "insolubility chasm" in Figure 3.4 are found with $Y < 0.25$.

The mechanisms of metal phytotoxicity are not understood completely, but a consensus does exist as to the importance of the following possibilities (ibid.): (1) displacement, by a nonessential metal (e.g., Cd), of an essential metal (e.g., Ca) bound to a bioligand; (2) complexation of a metal by a functional group in a biomolecule that effectively blocks the group from reacting further; and (3) modification, by interaction with a metal, of the conformation of a biomolecule that is critical to its biochemical function. All of these mechanisms are related to complex formation between a potentially toxic metal ion and the functional groups on a biomolecule. Indeed, they suggest that strong complex formers with biomolecules are likely to induce phytotoxicity.

The role of the Misono softness parameter in respect to these concepts can be appreciated after examining the phytotoxicity sequences in Table 3.3 (Sposito, 1986). For each class of plants, the ordering of metals from left to right reflects an increasing concentration of the metal (in M m^{-3}) required to produce a substantial toxic effect, with the smallest concentration associated with the most toxic metal. The sequences in Table 3.3 are remarkably similar. In the context of Table 3.2, they show that metal cations with the largest values of Y are the most toxic to plants. With the possible exception of Pb(II), "soft" metal cations—the strong complex formers—are most likely to participate in the three basic phytotoxicity mechanisms.

TABLE 3.3

Representative metal phytotoxicity sequences[a]

Organisms	Toxicity sequence[b]
Algae	Hg > Cu > Cd > Fe > Cr > Zn > Co > Mn
Fungi	Ag > Hg > Cu > Cd > Cr > Ni > Pb > Co > Zn > Fe
Flowering plants	Hg > Pb > Cu > Cd > Cr > Ni > Zn

[a]Based on data compiled by Nieboer and Richardson (1980) and by Eichenberger (1986).
[b]Hg = Hg(II), Fe = Fe(II), Cr = Cr(III), Co = Co(II), Mn = Mn(II), Pb = Pb(II).

From the point of view of atomic structure, then, essential metals can be characterized by ionic potentials lying outside the "hydrolysis chasm" ($30 < IP < 95\ nm^{-1}$) and by Misono parameters less than the "softness threshold", ($Y < 0.32$ nm). Metals for which $IP > 100\ nm^{-1}$ will exist as oxyanions in the soil solution (Mo), whereas metals for which $IP < 30\ nm^{-1}$ will exist as solvated cations (Mg, K, Ca, Mn, Fe, Cu, and Zn). Metals for which either $30 < IP < 95\ nm^{-1}$ or $Y > 0.25$ nm are potentially phytotoxic as aqueous species (e.g., Al(III), Ni(II), Hg(II), Cd(II)). Whether phytotoxicity is actually realized, however, depends on the speciation of the metal in the soil solution, as discussed in Section IV. A. (Bernhard et al., 1986).

VI. PESTICIDE MOBILITY IN SOILS

The pollutant organic compounds that react with soil are derived mainly from pesticides, fertilizers, and their degradation products (Sawhney and Brown, 1989). A consensus exists that the most important soil constituent that reacts with these pollutants is humus (ibid.; Schwarzenbach et al., 1993; Petruzzelli and Helfferich, 1993). Humus in solid form, either as a colloid or as a coating on mineral surfaces, can immobilize pollutants and, in some instances, detoxify them. Either colloidal or soluble humus (the fulvic acid fraction) can form strong complexes with organic compounds that then may travel freely with percolating water down into the soil profile. Pesticides that otherwise might remain near the land surface can be transported to groundwater by this mechanism (Sawhney and Brown, 1989).

Much of the molecular framework of soil organic matter is not electrically charged. This non-ionic structure can nevertheless react strongly with the uncharged part of an organic compound through van der Waals interactions (ibid.; Petruzzelli and Helfferich, 1993). The van der Waals interaction between non-ionic compounds, or non-ionic portions of compounds, and soil organic matter is often stronger than the interactions between these compounds and soil water (Buffle, 1988; Sawhney and Brown, 1989; Israelachvili, 1992; Petruzzelli and Helfferich, 1993; Schwarzenbach et al., 1993). Water molecules in the vicinity of large, nonpolar molecules are not attracted and so cannot orient their very polar OH bonds in ways that are compatible with the normal structure of liquid water (hydrophobic effect) (Israelachvili, 1992). The resultant disorder of this situation produces a low water solubility of the nonpolar molecule and a propensity for it to react with soil organic matter through van der Waals interactions (Sawhney and Brown, 1989). This reaction usually can be described by an effective distribution coefficient, K_{om}:

$$K_{om} = \frac{n_{om}}{[A(aq)]} \tag{13}$$

where n_{om} is the number of M of compound A that react with unit mass of soil organic matter in equilibrium with the soil solution concentration $[A(aq)]$ in M m^{-3} or M kg^{-1} of solution. Thus the units of K_{om} are m^3 kg^{-1} or kg kg^{-1}.

Equation 13 is analogous to Henry's law, in that it represents a partition of a compound between two phases, soil humus and the soil solution (ibid.). In the case of nonionic molecules, like the halogenated, aromatic rings in polychlorinated biphenyl (PCB) polymers or the organophosphates, this partitioning is expected to favor soil humus when van der Waals interactions and the hydrophobic effect are significant (ibid.; Schwarzenbach et al., 1993). For these compounds, the organic matter content is the soil property or chemical attribute that determines the amount of a pesticide or other organic pollutant adsorbed by the soil. Since the hydrophobic effect is inversely related to water solubility, it is reasonable to expect also that there will be an inverse relationship between K_{om} and water solubility. A negative correlation has often been observed statistically and is of the logarithmic form (Sawhney and Brown, 1989):

$$\log K_{om} = a - b \log S_W \tag{14}$$

where log is base 10 logarithm, S_W is the water solubility (g m^{-3}) of a compound whose partition coefficient is K_{om}, and a and b are empirical parameters. For example, with $a = 3.95$, $b = 0.62$, based on extensive measurements of K_{om} in the range 10–10^6 kg kg^{-1} correlated to S_W values in the range 10^{-3}–10^5 g m^{-3} (ibid.), Equation 14 predicts that atrazine, a herbicide with a water solubility of 33 g m^{-3}, will have a K_{om} of 10^3 kg kg^{-1}. On the other hand aldicarb, for which $S_W = 6 \times 10^3$ g m^{-3}, will have a K_{om} of only 41 kg kg^{-1}. Increasing water solubility corresponds to decreasing partitioning into soil organic matter.

VII. CONCLUSIONS

Soil chemical properties that have commonly been measured for agricultural soils, including organic matter content, pH, electrical conductivity, exchangeable sodium percentage, and cation exchange capacity, may be used as chemical attributes to assess soil quality. Soil mineralogy, although more difficult to determine than other soil chemical attributes, provides essential information about the ability of soils to retain nutrients in plant-available forms in some cases. Soil redox status, measured as pE, is dependent on other factors and is also not commonly measured in agricultural soils, although the availability, toxicity, or mobility of certain elements is highly dependent upon this parameter. The dependence of crop growth on these attributes is explained in terms of the speciation of elements in the soil, most importantly the distribution of the element between the solid and aqueous solution phases and also the forms of the element present in the solution phase, including complexes and the free ionic species. The overall speciation of an element in a soil may be influenced by several chemical processes. Solutes that do not interact with soil solids are highly mobile and susceptible to losses from the soil by leaching. Nutrient elements that are retained in the solid phase as a result of adsorption or precipitation reactions may be replenished into solution during the growing season by the release of adsorbed species or mineral dissolution. However, the formation of highly insoluble minerals or very strong complexes of a solute with soil particles may decrease the activity of that species in the soil solution to very low levels. In this case, the rate of transport of the element through the soil may be insufficient to supply the nutritional requirements of a plant, and a nutrient deficiency results. Certain species of plants have developed mechanisms to increase nutrient uptake under adverse soil chemical conditions. Many of these mechanisms are effective by inducing changes in the speciation of a nutrient among the solid and solution phases. The solubility and phytotoxicity of trace metal elements in the soil environment may be characterized based upon chemical indices and parameters that are related to atomic properties of these elements. Immobilization of phytotoxic elements and organic pollutants is favored by the formation of stable complexes between these species and soil particles. The speciation of these potentially hazardous elements and compounds in the soil is dependent upon the same soil chemical properties that influence the speciation of plant nutrients, although the specific mechanisms of reaction may be different.

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Chapter 4

BIOLOGICAL ATTRIBUTES OF SOIL QUALITY

E.G. GREGORICH, M.R. CARTER, J.W. DORAN, C.E. PANKHURST, and L.M. DWYER

I. INTRODUCTION

The biological attributes of soil include living organisms and material derived from living organisms. Living organisms (the *biotic* component of soil) include plants, animals, and microbes, ranging in size and function. After these organisms die, their residues remain in the soil in various stages of decomposition. Both living organisms and their residues interface with the *abiotic* component of the soil, which is non-living and chemically derived and includes minerals, clays, water, and chemical ions and compounds. These components are tightly linked, shaping and affecting each other.

Biological attributes of soil quality include the many soil components and processes related to organic matter cycling, such as total organic carbon and nitrogen, microbial biomass, mineralizable carbon and nitrogen, and light fraction; enzyme activities; and soil fauna and flora. These soil attributes are particularly fitting as indicators of soil quality, because they respond to change, both natural and human-induced. As well, plants have long been used as indicators of soil quality because their growth and visual appearance are regulated by a number of soil properties. In this chapter we describe these biological attributes and factors affecting them, including land management practices, and provide a rationale for their use in assessing soil quality.

II. SOIL ORGANIC MATTER

Soil organic matter, composed of various fractions, is a key attribute of soil quality (Fig. 4.1). It is the primary source of, and a temporary sink for, plant nutrients. It is important for soil quality because it helps to maintain soil tilth, aids the infiltration of air and water, promotes water retention, reduces erosion, and controls the efficacy and fate of pesticides. The dynamics of soil organic matter and its use as an attribute of soil quality are more fully addressed in Chapters 12 and 18.

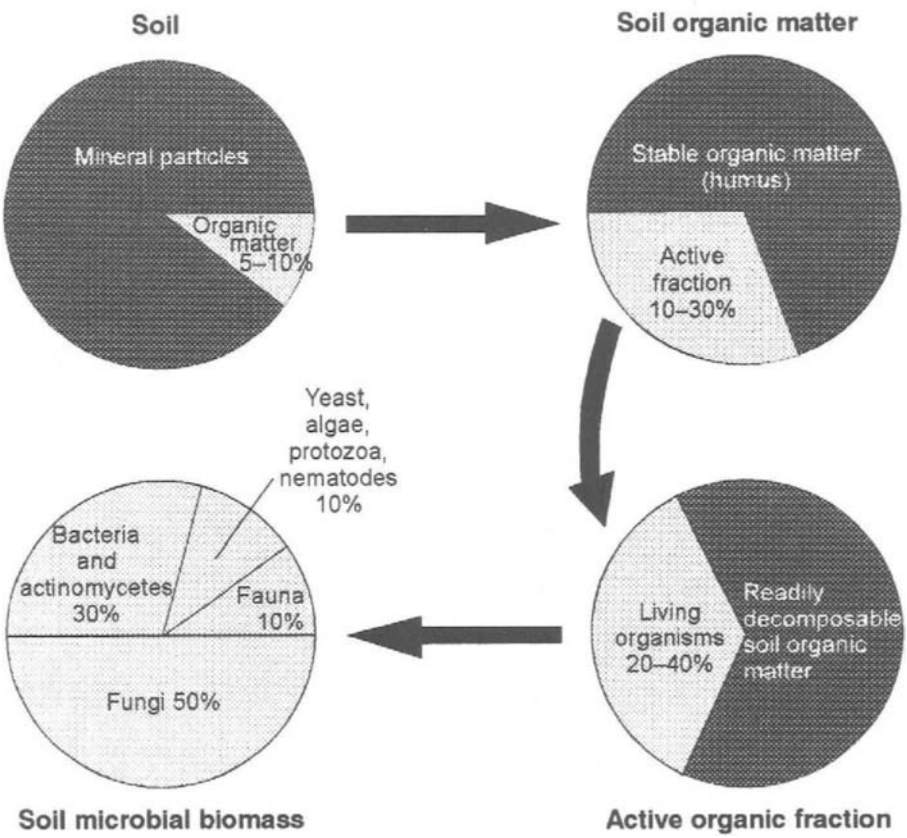

Fig. 4.1. Composition of soil organic matter, the active fraction, and the soil microbial biomass.

A. Total organic C and N

Soil organic matter is typically measured as soil organic C and N. The amount of organic C and N in a soil is a balance between how much is added to the soil (e.g., through the addition of organic residues and increased plant residues due to fertilization) and how much is removed (e.g., through microbial decomposition, mineralization, and erosion). The store of C in soil is important to the global C cycle as a medium-term pool.

1. Factors affecting total organic C and N

Organic matter levels decline when a soil is first tilled for two reasons. First, decomposition of the existing soil organic matter is enhanced because of changes in soil moisture, aeration, and temperature conditions, as well as the disruption of physically protected organic matter. Secondly, there is reduced replenishment of soil organic matter, because plant material is removed from the field and smaller amounts of plant (crop) residues are returned to the soil.

When a virgin soil is converted to agricultural production, C levels drop sharply during the first few years (even within two years in some cases; Davidson and Ackerman, 1993) following initial cultivation. The size and rate of loss of soil organic matter varies according to soil type. In some soils the loss of soil N may not be as large as the loss of soil C, because changes in N storage are dependent on management of N fertility (Ellert and Gregorich, 1996). Thus management-induced changes in soil C may not be estimated reliably from changes in N storage and a typical C:N ratio.

Management practices that maintain relatively high levels of soil organic matter include those that return relatively large amounts of residue to the soil and/or maintain soil environmental conditions that suppress decomposition (Odell et al., 1984; Liang and MacKenzie, 1992; Lal et al., 1994; Wander et al., 1994). For example, fertilization results in a more robust, abundant crop that contributes a greater volume of residues to the soil, resulting in higher levels of soil organic C. Continuous cropping and tillage practices that maintain residue cover also contribute greater organic inputs to the soil and result in increased soil organic C content.

Studies on the long-term effects of residue management in wheat systems showed that levels of organic C and N declined except in soils receiving large inputs of manure (Rasmussen and Parton, 1994). However, the decline of organic C and N was less in soils to which pea vines had been added along with straw residues, than in soils that only received straw alone. Simulation modelling of these data showed that soil C changes were linearly related to above-ground C inputs and that stabilization of C was highest in soils with high fertilizer additions (Parton and Rasmussen, 1994).

2. Total organic C and N as indicators of soil quality

Soil organic matter has been suggested as the single most important indicator of soil quality and productivity (Larson and Pierce, 1991; Doran and Parkin, 1994;

1996). As with any indicator, determining an appropriate baseline or reference soil is critical to the assessment of soil quality. Changes in soil organic matter are usually assessed by comparing sites subjected to specific agricultural practices with reference sites, such as native forest or grassland. Comparisons of total organic C and N, for example, have been made between native forest or grassland soils and their cultivated counterparts, as well as between soils under different crop rotations, fertilization regimes, and tillage treatments.

Large perturbations to the soil, such as the conversion of native forest or grassland soil to agricultural production, are reflected in large changes in the total mass of organic C and N. Within agricultural systems, however, changes in total organic C and N may be more subtle (Janzen et al., this volume). The amount of C or N added to soil annually is small (usually < 10%), relative to the amount present in the soil so changes in the total mass of organic C and N may be difficult to detect. Therefore, extended periods of time are required to observe significant changes in soil organic matter, emphasizing the utility and need for long-term field experiments (Christensen and Johnston, this volume; Campbell et al., this volume).

Although soil organic matter is recognized as a critical component of soil quality, comparing the masses of organic C and N may not provide an adequate assessment of the important changes that occur in the soil organic matter (Wander et al., 1994). Nevertheless, soil organic matter content can be used as a coarse measure of soil quality, because high levels usually show good correlations with other desirable attributes of soil, such as high levels of microbial biomass, available plant nutrients, and good soil structure (Pankhurst, 1994).

B. Light fraction

The light fraction (LF) is mainly composed of plant residues but also contains animal and microbial residues in various stages of decomposition. It is a transitory pool of organic matter between fresh plant residues and humified organic matter; thus it is highly enriched in carbon and nutrients. Although the LF represents only a small portion of the soil mass, it is an important attribute of soil quality because its short turnover time makes it an important carbon substrate and source of nutrients.

1. Factors affecting the light fraction

The composition of the LF is controlled by the chemical composition of plant material and other residue inputs, as well as by factors affecting the decomposition process, such as type of decomposer organisms and soil environmental conditions. In cultivated soils, the LF typically contains 20–30% C and 0.5–2.0% N (Dalal and Mayer, 1986; Janzen et al., 1992) and makes up 2–18% of the total C and 1–16% of the total N found in the whole soil (Gregorich and Janzen, 1996). The C:N ratio of the LF is usually between that of whole soil and plant tissue, because LF organic matter is less humified than the organic matter found in the whole soil (Greenland and Ford, 1964) and undergoes more loss of C than N in the initial stages of decomposition (Adams, 1980).

The LF is largest when decomposition rates are lowest, such as under arid and cold conditions (Janzen et al., 1992). The proportion of C and N in the LF tends to increase with increasing latitude (Christensen, 1992), a reflection of differences in both vegetation and climate. The amount of LF may also vary seasonally; Spycher et al. (1983) found that it increased by 50–100% from early spring to summer. Levels of LF are usually greatest in the top few cm of the soil, although its distribution is more uniform in intensively tilled soils (Biederbeck et al., 1994).

The LF is a likely habitat for microorganisms and the site of intense decomposer activity because of its high levels of C and N. The microbial biomass is thought to make up a significant portion of the LF (Ladd et al., 1977), and microscopic examinations show that fungal hyphae and faunal debris are significant constituents of the LF (Spycher et al., 1983). Thus, the composition and size of the microbial community has an effect on the composition and amount of the LF.

2. Effects of management practices on LF

The LF accounts for a higher proportion of total organic C in virgin soils than in cultivated soils (Skjemstad et al., 1986). Initial conversion of virgin lands to agriculture typically results in a disproportionately large loss of LF organic matter (Dalal and Mayer, 1986). Dalal and Mayer (ibid., 1987) observed that the rate of loss is greatest in fine-textured soils.

Agricultural practices that affect the amount of residue input into soil or the rate of residue decomposition will affect the LF, which has relatively little stability against decomposition in the soil. Levels of LF C and N decrease with increased use of fallow (Janzen, 1987). On average, the LF carbon content of soils under frequent summerfallow is about half that of soils under continuous cropping (Bremer et al., 1994; Biederbeck et al., 1994). Fallowed soils have negligible plant production and may have soil moisture and temperature conditions that enhance decomposition of free organic matter (Janzen et al., 1992).

The LF is also affected by cropping practices. Cropping to grasses or legumes typically enhances LF organic matter levels, as does continuous pasture (Angers and Mehuys, 1990; Conti et al., 1992). The amount of LF was found to be higher in a rotation of short-term leguminous hay with annual crops, such as wheat, than in fallow–wheat systems (Janzen et al., 1992), and lower in a rotation of wheat with lentils than in a continuous wheat system (Biederbeck et al., 1994).

Addition of nutritive amendments can increase plant biomass production and crop residue inputs to the soil, thereby increasing LF and soil C contents. In a comparison of fertilized and non-fertilized soils, Gregorich et al. (1996a) observed that the LF accounted for a significant part of the gain in total soil C, and that the gain was attributed to recent residue additions (Fig. 4.2). Manure applications also increase LF organic matter. Long-term application of low rates of animal manure resulted in a nearly two-fold increase in LF C in a fallow–wheat–wheat system (Bremer et al., 1994).

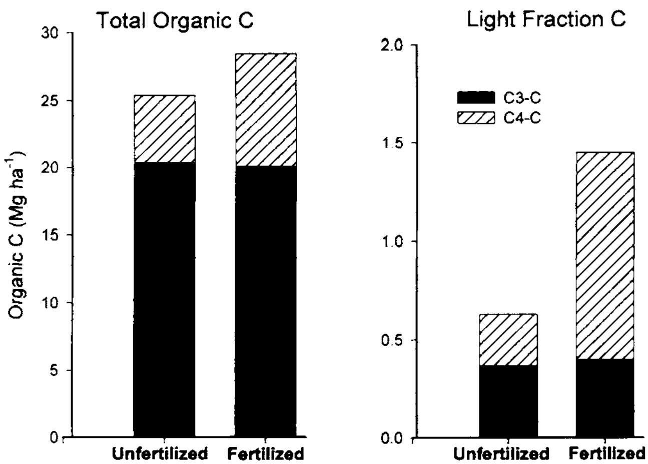

Fig. 4.2. Amounts of total organic C and maize-derived C in the whole soil and light fraction in the topsoils of fertilized and unfertilized soils after 32 years under maize (data from Gregorich et al., 1996a).

3. Light fraction as an indicator of soil quality

The LF has a relatively rapid turnover (Gregorich et al., 1996a; 1997) and is an important source of labile C, making it a sensitive indicator of soil quality, particularly organic matter quality. This is not surprising, because LF is much more responsive than total organic matter to changes in management. For example, a long-term study of wheat systems showed that the range of LF C among treatments was about 15 times greater than that of total organic C (Biederbeck et al., 1994), a finding confirmed by others (Janzen et al., 1992; Bremer et al., 1994). However, although the sensitivity of the LF to management effects is well established, the usefulness of the LF to predict changes in organic matter is not known. Increased LF generally signals increased levels of labile organic matter, but the eventual conversion of that transient C to stable organic matter has not been described adequately.

C. Macroorganic matter

The macroorganic matter is that fraction associated with sand sized-particles, and separated from the soil by sieving. Generally, it consists mainly of fine root fragments and other organic debris (Cambardella and Elliot, 1992). Macroorganic matter is generally a much larger proportion of the soil C than the LF and can account for well over 10% of the soil C (Carter et al., 1994; Gregorich et al., 1994). Chemical analysis has shown that macroorganic matter is more decomposed than LF (Gregorich et al., 1996b) and, like the LF, is very sensitive to soil cultivation and changes in cropping practices. Dynamics of the macroorganic C (separated using a 50-µm sieve) were found to relate well with the "plant structural material" compartments used in the Rothamsted and Century C models (Balesdent, 1996).

1. Effects of management on macroorganic matter

Macroorganic matter decomposes more quickly than whole soil or particle fractions, because it is generally free of mineral particles and the protection from decomposition that this association may impart. Therefore it responds to changes in management practices that affect soil environmental factors. Cambardella and Elliott (1992) found that soils under no-till had higher concentrations of particulate organic matter than those under stubble mulch and bare fallow treatments with comparable residue inputs, probably because residues on the soil surface are more desiccated and thus decompose more slowly. Angers et al. (1993b) found that reduced tillage can maintain or increase the level of macroorganic organic matter relative to that under conventional tillage.

2. Macroorganic matter as an indicator of soil quality

Relative to whole soil and other particle size fractions, macroorganic matter has a rapid turnover time (Balesdant, 1996). Compared to the LF, macroorganic matter contains more aromatic C, is more decomposed, and has a longer turnover time in soil (Gregorich et al., 1996b). Like the LF, macroorganic matter appears to be a sensitive indicator of changes in total soil organic matter. As with other labile fractions of organic matter, the reliability of macroorganic matter as an indicator of soil quality depends on the mechanism of soil organic matter accumulation (Janzen et al., 1997). If the change in macroorganic matter occurs because of greater C inputs to the soil, then it will eventually be reflected in higher levels of soil C. On the other hand, if the change in macroorganic matter occurs in response to the suppression of the rate of decomposition, then the increased macroorganic matter may not indicate increased organic matter beyond that accumulated in the macroorganic matter itself.

III. SOIL BIOTA

A. Microbial biomass

The soil microbial biomass is the living microbial component of the soil, comprising mainly bacteria and fungi but also including soil microfauna and algae (Fig. 4.1). Although the soil microbial biomass accounts for only 1–3% of the organic C and 2–6% of the organic N in soil (Jenkinson, 1987), it plays a key role in soil organic matter dynamics. It controls the transformation of organic matter in soil and influences C storage, and is both a sink (during immobilization) and temporary source (during mineralization) of plant nutrients. The diverse metabolic activities of soil microorganisms, particularly bacteria, regulate the energy and nutrient cycling that take place in soil and are important in the global cycling of many inorganic compounds, particularly N, S, and P.

1. Factors affecting microbial biomass

The amounts of C and N in the microbial biomass are usually closely related to amounts in the mineralizable fraction of soil organic matter (Paul and Voroney, 1984). Anderson and Domsch (1989) found that the relationship between microbial biomass and soil organic C was strongest in soils with less than 2.5% organic C. In some cases the amount of soil organic C in the microbial biomass is influenced by the amount of soil N (Beare et al., 1990).

Low microbial biomass is often associated with soils of low pH (Vance et al., 1987) and can be enhanced by additions of lime to the soil (Carter, 1986). High concentrations of toxic substances, such as heavy metals, may reduce microbial biomass levels (Brookes et al., 1986). Agrochemicals, including herbicides, may temporarily affect the soil microbial biomass, but probably only at levels that exceed those found in actual field conditions (Wardle and Rahman, 1992).

The microbial biomass shows some seasonal variation associated with climatic factors. It can be lost quite quickly when soil dries out, but recovers, sometimes very quickly, upon rewetting (Sparling et al., 1986). During this cycle, a significant portion of microbial biomass C and N may be released, which in turn may have impacts on crop production and groundwater quality.

2. Effects of management practices on microbial biomass

Because soil is usually a C-limited environment, microbial biomass responds quickly to organic inputs, and its size usually increases with the addition of readily hydrolysable C sources to the soil (Ocio et al., 1991). This suggests that the microbial biomass can be used as an indicator of changes in soil C (Powlson et al., 1987). However, this may not be valid for soils with relatively low total C contents (Wardle, 1992). The effects of N amendments on soil biomass are inconsistent, because in some cases microbial biomass increases and activity is stimulated; in other cases there is the opposite or no effect (ibid.).

Tilled soils contain less microbial biomass than soils under no-till, reduced tillage, and perennial crops (Smith and Paul, 1990). Reduced tillage systems result in increased structural (aggregate) stability, which is also associated with an increased microbial biomass (Gupta and Germida, 1988; Drury et al., 1991). Networks of fungal hyphae are more easily established in soil that is minimally disturbed than in soil under regular tillage, and the microflora of soil under reduced tillage is generally dominated by fungi (Holland and Coleman, 1987).

3. Microbial biomass as an indicator of soil quality

Because of its high turnover rate relative to the total soil organic matter, the microbial biomass can quickly respond to changes in soil processes resulting from changes in management. The microbial biomass C can be divided by total organic C (Anderson and Domsch, 1989) or CO_2—C respired (Visser and Parkinson, 1992) in order to make comparisons between soils under different managements having different organic matter contents. Changes in the ratios generally reflect both organic matter inputs into and outputs from the soil and conversion of organic matter to microbial biomass C. The ratio of microbial biomass C to total organic C has been useful to elucidate changes in organic matter under different cropping (Anderson and Domsch, 1989) or tillage (Carter, 1991) systems, as well as in soil polluted by heavy metals (Brookes, 1995).

Sparling (1997) suggested that, because the "ideal" microbial biomass content for a healthy soil has not been defined, it is crucial that a soil-specific baseline be used for comparisons. The "target" value can be derived from the same soil type under alternative land management (Fig. 4.3).

A study in Ontario, Canada was initiated to evaluate the loss of soil organic matter under different managements. Soils under annual cropping were compared to soils under perennial grass and/or forest (Table 4.1). Baseline values for soil C, microbial biomass C, and the ratio of microbial C to soil C were different for each soil type. With reference to these baseline values, soils that were more intensively managed lost soil organic matter, but the microbial biomass C pools declined much faster than the total soil C, and the ratio of microbial C to soil C also decreased.

B. *Mycorrhiza*

Mycorrhizae are ubiquitous symbiotic fungal associations with plant roots that have several functions. They aid in the uptake from soil of relatively immobile nutrients, such as P and Zn, may help plants withstand environmental stress (drought, salinity, heavy metals), and reduce attack from root pathogens (Fitter, 1989). There are two types of mycorrhizae—endomycorrhiza, which include the vesicular–arbuscular mycorrhiza (VA mycorrhiza), and ectomycorrhiza, which are usually found on woody plant species. VA mycorrhiza are present in most agricultural soils. Most species have the capacity to infect a range of plants, but they may differ in their effectiveness in promoting nutrient uptake by plants

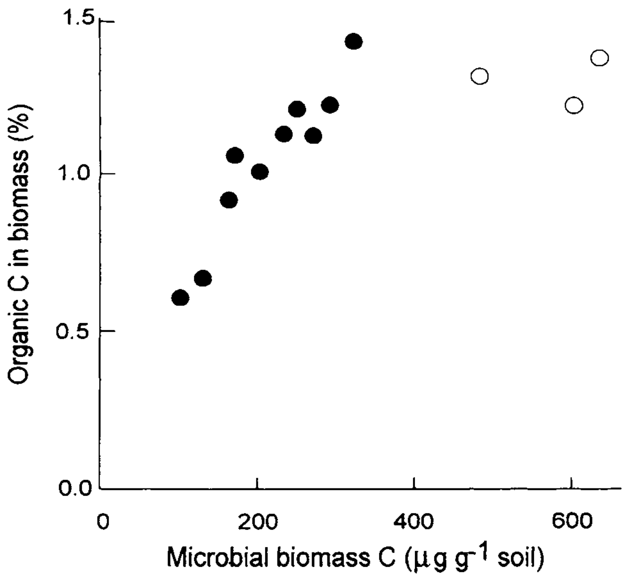

Fig. 4.3. The ratio of microbial biomass C to total organic C in different cropping systems (●) compared to a permanent grassland reference area (○) (data from Carter, 1991).

TABLE 4.1

Soil and microbial biomass C, and the ratio of microbial biomass C to soil C in the surface of soils under different managements in Ontario, Canada (data from Ellert and Gregorich, unpublished)

Site and Management	Total C (%)	Microbial biomass C ($\mu g\ g^{-1}$)	Mic. biom. C/Soil C (%)
Woodslee			
Corn (fertilized)	2.2	219	1.02
Corn (not fertilized)	2.0	207	1.06
Perennial grass (fertilized)	5.1	1464	2.85
Perennial grass (not fertilized)	4.5	1235	2.74
Forest	7.8	2970	3.80
Vineland			
Peach orchard & annual ryegrass	1.3	364	2.80
Vegetable crop (minimum tillage)	1.5	458	3.05
Cherry orchard & perennial grass	2.3	893	3.88
Kemptville			
Cropland (stover removed for 20 yr)	1.8	126	0.70
Cropland (residues returned)	1.7	155	0.91
Forest	3.6	946	2.63
Plainfield			
Cropland (tilled annually)	2.1	367	1.75
Perennial pasture	4.6	1173	2.55
Forest	6.1	1689	2.77

(Bethlenfalvay, 1992; Jakobsen, 1994), which may provide options for their management. The importance of VA mycorrhiza in sustainable agriculture is based on their link between plant and soil. As agents of nutrient transport between plant and soil, they affect soil conservation, soil fertility, and plant nutrition (Bethlenfalvay, 1992).

1. Factors affecting VA mycorrhiza

Species of VA mycorrhiza differ in their response to soil and environmental factors, although there is little published evidence of quantitative associations between soil properties and the quantities of VA mycorrhiza in the soil (Abbott and Robson, 1994). The geographic distribution of some fungi corresponds to soil characteristics such as pH (Robson and Abbott, 1989) and possibly temperature (Schenck and Smith, 1982). Adverse soil conditions, such as high salinity or the presence of heavy metals, can have varying effects on the abundance and activity of VA mycorrhizal fungi in the soil (Bethlenfalvay, 1992). In some circumstances the fungi may protect the plant from such stresses, whereas in others the fungi may be adversely affected.

Perhaps the single most important factor affecting VA mycorrhizal fungi is the presence or absence of a suitable host plant. Because these fungi are obligate plant symbionts, cropping sequences influence their populations. Mycorrhizal inoculum density declines when soils are kept fallow for extensive periods of time (Thompson,

1987) or when non-mycorrhizal plants, such as cruciferous crops, are grown. This may have a significant effect on subsequent crop production and was the cause of the phenomenon "long fallow disorder" in vertisolic soils in northern Australia (Thompson 1987, 1991). Thompson (1991) showed that densities of VA mycorrhiza spores before sowing, per cent colonization, and yields were inversely related to the length of time that the soil was left in fallow (Fig. 4.4a,b,c). Furthermore, yield response to P and Zn fertilization was positively related to fallow length, strongly suggesting that mycorrhizae are involved in this effect (Fig. 4.4d). In other studies, plant species influenced the species composition of VA mycorrhizal communities (Johnson et al., 1991), indicating that cropping sequence may also provide an effective means of manipulating the composition of mycorrhizal populations.

2. Effect of management practices on VA mycorrhiza

The effects of agricultural management practices on the abundance and activity of VA mycorrhizal fungi are complex (see reviews by Johnson and Pfleger, 1992; Thompson, 1994; Abbott and Robson, 1994). Thompson (1994) concluded that management practices that ensure high levels of inoculum of VA mycorrhizal fungi

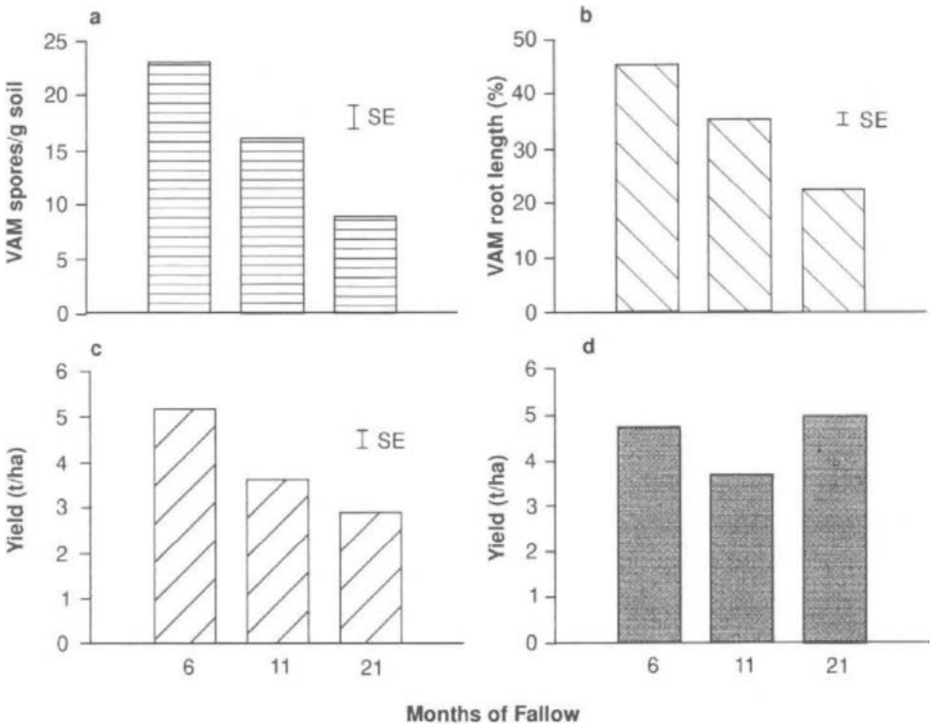

Fig. 4.4. Effect of fallow duration on soil densities of (a) VAM fungal spores, (b) mycorrhizal colonization of corn roots, (c) grain yield of corn without, and (d) with P and Zn fertilizer (data adapted from Thompson, 1991).

in soil include cropping with hosts that become extensively colonized, selective use of fungicides and fertilizers, reduced use of fallows, and limited topsoil disturbance.

Although fallow and cropping sequence, discussed above, can significantly affect the abundance and composition of VA mycorrhiza populations in the soil, no simple generalizations can be made about the effects of tillage, fertilizer, or pesticides on mycorrhizae. Tillage disrupts soil networks of mycorrhizal hyphae and may impair nutrient uptake and reduce crop yields (Evans and Miller, 1990). Fertilizers have been reported to both increase and decrease quantities of mycorrhizae. Such contradictions are not surprising considering the complexity of mycorrhizal systems and the myriad of undefined variables and interactions inherent in field research (Johnson and Pfleger, 1992). Similar difficulties are associated with defining the effects of pesticides on mycorrhizae. Formulation of the pesticide, application rate, timing of application, soil type, environmental conditions, and identity of the crop and VA mycorrhizal fungi within the system may mediate the effects pesticides have on mycorrhizae (Johnson and Pfleger, 1992).

3. VA mycorrhizae as indicators of soil quality

In all ecosystems, the VA mycorrhizal fungi play a crucial role in sustainable plant productivity and maintenance of soil structure, because they link plant functions with soil functions by acting as mediators of nutrient exchange between them. Thus there is a closed chain of cause-and-effect relationships in the role of VA mycorrhizal fungi in the plant–soil system: the fungi improve the health and development of their host by enhancing plant nutrition and resistance to disease and stress; the more vigorous plant is a better source of C to the soil, which encourages the activity of the soil biota; the products of microbial metabolism enhance soil aggregation; and better soil structure permits better plant and VA mycorrhiza-fungal growth. These relationships were validated by demonstrating strong positive correlations between the abundance of selected VA mycorrhizal fungi in a red-brown earth (calcic rhodoxeralf) and other soil properties (aggregate stability, microbial biomass, cellulose decomposition activity, CO_2—C respiration, and protease activity) (Pankhurst, unpublished), and management practices associated with increased soil quality (reduced tillage, reduced use of fallow and cropping sequences; Thompson, 1994). Results reported by Bethlenfalvay and Barea (1994) indicated that VA mycorrhizal fungi influenced the carbon allocation between plant (measured as seed yield) and the soil (measured as the formation of water-stable aggregates). They concluded that mycorrhizal fungi affect two biologically controlled aspects of sustainable agriculture: crop production and soil quality.

C. Fauna

Soil fauna comprise a large variety of organisms, grouped according to their size as microfauna (<100 μm in width; e.g., protozoa, nematodes), mesofauna (100–2000 μm in width; e.g., collembola, mites, enchytraeids), and macrofauna (>2000 μm in width; e.g., earthworms, termites). Collectively these organisms are major determinants of soil processes affecting the fertility and structure of soils. They contribute substantially to a soil's resilience and capacity to support plant growth (Crossley et al., 1989; Lee and Foster, 1991; Curry and Good, 1992). Therefore some measure of soil faunal abundance, diversity, or activity may provide a useful indicator of soil quality.

Soil microfauna are essentially aquatic organisms that exist in water films on particle surfaces in the soil. Because of their small size, they have limited ability to modify soil structure directly. However, they affect soil nutrient availability through their trophic interactions with soil microorganisms (Ingham et al., 1985). Protozoa and certain free-living nematodes consume soil bacteria and fungi. Depending on their feeding intensity, they can significantly affect microbial numbers and hence influence the rate of turnover of microbial biomass. This in turn affects organic matter mineralization and nutrient availability.

Soil mesofauna exhibit a variety of feeding strategies and functional roles in soil processes. Collembola and mites have a major role in regulating fungal and microfaunal populations and a facilitory role in the fragmentation of organic residues. The larger and more active mesofauna (e.g., enchytraeids) have a major role in the fragmentation of residues and also affect soil porosity through burrowing activities (van Vliet et al., 1993) and aggregation via production of faecal pellets (Lee and Foster, 1991).

Soil macrofauna are the most conspicuous soil animals and have the greatest potential for direct effects on soil functional properties. The macrofauna include ants, termites, millipedes, adult and larval insects, earthworms, snails, and slugs. They comminute and redistribute organic residues in the soil profile, increasing the surface area availability of organic substrates for microbial activity. Certain groups, particularly the ants, termites, and earthworms, can substantially modify soil structure through the formation of macropores and aggregates (Lee and Foster, 1991). These effects may influence water infiltration and solute leaching through soils (Fraser, 1994), and hence the soil's capacity to function as an environmental buffer.

1. Factors affecting soil fauna

Soil moisture, temperature, and the availability of food are the primary environmental and biotic factors that affect all soil fauna populations (Gupta, 1994; Heal and Dighton, 1985). For example, detritus inputs and the stability of the microhabitat (Heal and Dighton, ibid.) are important factors governing the abundance and diversity of soil mesofaunal populations. Habitat stability is also important for soil macrofauna.

2. Effects of management practices on the soil fauna

Soil and crop management practices (tillage, crop rotations, plant residue retention) markedly affect the composition and abundance of soil faunal communities (Crossley et al., 1989; Curry and Good, 1992; Lavelle et al., 1994). The effects are largely mediated through changes in the physical, chemical, and biological status of the soil. In some soils, for example, tillage coupled with incorporation of plant residues creates a soil environment that favours a bacteria-dominated decomposer community with the capacity for rapid decomposition of organic residues and release of nutrients. This in turn favours the development of a soil faunal population dominated by bacterivorus protozoa, bacterivorus nematodes, and enchytraeids (Hendrix et al., 1986). In contrast, reduced tillage with surface retention of organic residues favours a fungi-dominated decomposer community with slower nutrient turnover. This in turn favours the development of a soil fauna dominated by fungivorus nematodes, collembola, and earthworms (ibid.). These linkages or trophic interactions between micro-, meso- and macrofaunal groups are conceptualized as food webs in the soil (Ingham et al., 1985; Hunt et al., 1987; Beare et al., 1992). The structure of the soil food web reflects the direction, and influences the rate, of nutrient and energy flows through the system and may be useful as an indicator of management-induced change in soil quality (Elliott, 1994; Linden et al., 1994).

Other more specific management practices, such as fertilizer or pesticide application, also have significant effects, positive or negative, on the abundance and diversity of soil fauna. Elliott and Coleman (1977) and Griffiths (1990) reported a significant increase in active protozoan populations following N fertilization, whereas Solhenius (1990) observed a reduction in protozoan biomass following application of high levels of inorganic fertilizer (>120 kg N ha^{-1}). Similar increases or decreases in collembolan abundance (Wiggins et al., 1979), nematode abundance (Solhenius, 1990), and mite abundance (Siepel and van de Bund, 1988) in response to N fertilization have been reported. The reasons for these differing responses are complex and reside in differences in the cropping systems being examined, the time between fertilizer application and sampling, and other physiochemical factors. The effects of pesticides on micro- and meso-fauna are generally negative but short-term (Gupta, 1994).

Fertilizer and pesticide applications to soils can also have significant effects on soil macrofauna. Generally speaking, where fertilizer applications increase plant growth the effect on soil macrofauna, notably earthworms, is also favourable (Fraser, 1994). However, earthworms are highly sensitive to soil pH, so that the repeated addition of ammonia-based fertilizers to poorly buffered soils may harm earthworms (ibid.). Earthworms are sensitive to many insecticides and fungicides but are relatively insensitive to most of the commonly used herbicides (Lee, 1985).

Management of crop residues using different tillage practices affects the microhabitat and hence number of soil fauna. Winter et al. (1990) found that microarthopods (Collembola and Acarina) in the surface 5 cm of soil were more concentrated in soil under long-term no-tillage than conventional tillage (Table 4.2). But when examined over a depth of 15 cm, the size of the microarthropod populations did not differ between tillage systems. However, production of bromegrass for 3–4 years following conventional tillage significantly increased the number of microarthopods.

3. Soil fauna as indicators of soil quality

There have been several documented attempts to use soil fauna as indicators of soil quality, especially in relation to soil contamination with industrial chemicals and heavy metals (Foissner, 1987; Paoletti et al., 1991). For example, Koehler (1992)

TABLE 4.2

Mean number of microarthropods (Collembola and Acarina) in soil under long-term (19 yr) conventional and no-tillage maize cropping or conventional tillage followed by 3–4 yr of bromegrass (data from Winter et al., 1990.)

Soil Depth (cm)	Conventional tillage	No-tillage	Bromegrass
		Mean number (1000 m^2)	
1987 0–5	5.78 *c*	12.42 *b*	15.90 *a*
1988 0–5	12.53 *c*	13.91 *b*	16.24 *a*
5–10	9.58 *b*	9.52 *b*	14.82 *a*
10–15	12.63 *b*	9.19 *b*	12.23 *a*

Means within a soil depth followed by the same letter are not significantly different ($P < 0.05$).

showed significant effects of toxic chemicals on populations of collembola and mites and reported that the successional reestablishment of populations showed the effects of the chemical for several months after it could no longer be detected in the soil. Similarly, shifts in individual species and assemblages of micro- and meso-fauna have been demonstrated in soils contaminated with heavy metals (Paoletti et al., 1991; Ohtonen et al., 1992) and radioactive fallout (Krivolutzkii and Pokarzhevskii, 1991). Other studies have examined the bioaccumulation of heavy metals and pesticides in the tissues of collembola and mites (van Straalen et al., 1987) and earthworms (Lee, 1985) as a potential indicator of the level of soil contamination.

Linden et al. (1994) suggest that soil fauna can be used as indicators of agricultural soil quality at three different levels: individual organisms or populations; communities, including diversity and trophic associations; and mediators of biological processes. At the organism level, changes in abundance, activity, physiology, or growth rate are some measures with good diagnostic potential. It is important, however, to show that such changes result from changes in the soil and not from natural temporal fluctuations of the organism (Curry and Good, 1992; Koehler, 1992). At the community level, responses of the soil fauna reflect the underlying structure of the food web. Food-web structure in itself could be used as an effective indicator, but the effort required to obtain such information precludes its practical application (Beare et al., 1992).

Influences of soil fauna on soil biological processes may offer the most valuable long-term indicators of soil quality. The disappearance, decomposition, and release of nutrients from crop and animal remains, and the development of biopores and mixing of organic and mineral soil components are two of the major functional properties of soils that directly involve soil fauna. The soil fauna involved, the processes, and the products have all been used to indicate the performance quality of the soil (Wolters, 1991). Earthworms are highly involved in these processes, and their presence in soils is usually regarded as a sign of high soil quality. However, although earthworm activity in the soil environment is desirable, it is not critical; many highly productive soils do not contain earthworms.

IV. MICROBIAL PROCESSES

A. Mineralizable C and N

Carbon mineralization is usually measured as the gross flux of CO_2 from soil during an incubation and thus indicates the total metabolic activity of heterotrophic soil organisms. Nitrogen mineralization is usually determined as the net flux of inorganic N during a soil incubation and represents the balance between gross mineralization and immobilization by soil organisms. Mineralizable C and N are usually measured in laboratory incubations of soil, although field incubations are commonly used to assess decomposition under realistic environmental conditions (Anderson, 1982; Raison et al., 1987).

1. Factors affecting mineralizable C and N

Mineralization of organic material in terrestrial ecosystems is regulated by three types of factors: organisms, the physical/chemical/climatic environment, and the quality of the plant residue (Fig. 4.5; Swift et al., 1979). Climatic variables, such as temperature and moisture, have a fundamental influence, but these factors are interactive, and particular factors may be more or less significant at different times and under different circumstances. For example, soil pH affects the activity of decomposer organisms. The quality of plant residues, as characterized by the chemical constituents of the plant material, may stimulate or inhibit microbial activity or feeding by soil fauna.

A number of factors affect the amount of organic matter mineralized under laboratory conditions. The quantities of mineralizable C or N are dependent on bioassay conditions, including temperature, moisture, aeration, the duration and measurement interval of the incubation, and the pre-treatment of the sample (especially drying).

2. Effects of management practices on mineralizable C and N

Carter and Rennie (1982) found significantly greater amounts of mineralizable C and N in zero-tilled soils than in conventionally tilled soils because of the effects of crop residues concentrated at the soil surface under no-till. Carbon and N mineralization are strongly related to the frequency of fallow in the rotation. Cumulative C and N mineralization were approximately two times greater in soils continuously cropped to wheat compared to soils under fallow–wheat rotation (Biederbeck et al., 1994). Fallowing not only hastens the loss of mineralizable N but

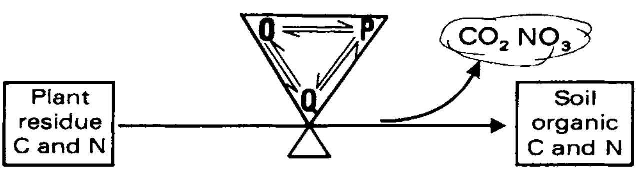

Fig. 4.5. The control of mineralization by organisms (O), the physical/chemical/climatic environment (P), and the quality of plant residue (Q) (adapted from Swift et al., 1979).

also tends to promote the conversion of labile N materials into stable, humified organic materials (Janzen, 1987).

Soils under longer rotations receiving adequate amounts of fertilizer usually have relatively greater amounts of mineralizable N. Biederbeck et al. (1984) reported that mineralizable N in adequately fertilized continuous wheat was 45% higher than in soil under fallow–wheat and 17% higher than in fallow–wheat–wheat. The positive effect of fertilization on mineralizable N was shown by the observation that soil receiving N and P had 25% more potentially mineralizable N than soil receiving only P. In general, N mineralization reflects C mineralization, but differences in N mineralization between management treatments are usually smaller (Biederbeck et al., 1994). The amounts of mineralizable C and N immediately following plowdown of a forage were found to be significantly higher than prior to establishment of the forage, demonstrating the labile nature of this organic matter (Janzen, 1987).

3. Mineralizable C and N as indicators of soil quality

Interpreting mineralizable C and N as measures of soil quality requires that laboratory bioassay results be related to transformations occurring in situ. Often the moisture and temperature constraints that occur in the field are removed in laboratory incubations. Thus laboratory measurements of mineralizable C and N represent maximum potential rates that may be obtained only rarely in the field.

Carbon and N mineralization are good estimates of labile organic matter because they provide a direct measure of organic matter turnover. Carbon mineralization can be used to assess the decomposition or persistence of organic wastes or crop residues and the contribution of soil organic matter to atmospheric CO_2. Mineralizable N is an important attribute of soil quality, because it can be used to assess the capacity of soil organic matter to supply inorganic N to the growing crop. The mobile compounds released during incubations may function as plant nutrients, water pollutants, precursors for stable organic matter, or precursors for gases released to the atmosphere. For example, the main component of inorganic N, NO_3, is mobile and readily lost through denitrification and leaching through the soil. Thus the measurement of mineralizable N may also have important implications for the health of atmospheric and aquatic ecosystems.

Sparling (1997) considers mineralizable N to be a useful integration of chemical, physical, and biological aspects of soil quality, because it combines both the accumulation of N through previous biological activity, the present organic matter status of the soil, and the current N mineralization activity of the soil microorganisms. Biederbeck et al. (1997) rated the response of several labile fractions of organic matter to green manuring and found that the initial potential rates of N mineralization and C mineralization were more sensitive than biomass C and light fraction soil organic matter.

B. Carbohydrates

Carbohydrates, originating from microbial and plant sources, account for 5–25% of the total C content in soil. They are a labile form of C and an energy source for microorganisms, and those derived from microbial sources are important binding agents for aggregation (Cheshire, 1979). Carbohydrates influence soil quality mainly through their role in the formation and stabilization of aggregates which promote water infiltration.

1. Effect of management practices carbohydrates

No-till management usually increases the carbohydrate content of soils (Arshad et al., 1990; Hu et al., 1995), and this change can occur in a short period of time. Angers et al. (1993a) reported that the ratio of carbohydrate C to total C was greater in no-tilled soil than in moldboard-plowed soil after only three cropping seasons. Perennial grass cropping has been shown to induce increases in carbohydrates within a similar period of time (Chantigny et al., 1997). Roberson et al. (1995) observed that a vetch winter cover crop performed better than a non-N-fixing crop or moderate N fertilization in stimulating production of microbial carbohydrates and improving soil structure.

2. Carbohydrates as an indicator of soil quality

Changes in management sometimes improve soil structure without any detectable change in total C or total carbohydrate content (Baldock et al., 1987), suggesting that a particular fraction is responsible for the structural changes. Thus, research on carbohydrates has focussed on the origin and relative contribution of individual monosaccharides to the total carbohydrate pool. Ratios of monosaccharides (e.g., mannose and xylose) are often used to identify the source of carbohydrates (Oades, 1984). Soil carbohydrate ratios, in combination with information on microbial biomass, have been suggested as useful indicator of changes in soil organic matter as a function of biotic and management regimes (Hu et al., 1995). Roberson et al. (1995) isolated the heavy fraction (which was enriched in microbial carbohydrates) to assess rapid changes in soil structure and observed significant correlations between heavy fraction carbohydrate content and both aggregate stability and K_{sat}. Different extraction techniques, using hot water or dilute acid, have been used to isolate labile carbohydrates, which are believed to be involved in stabilizing aggregates (Angers et al., 1993a).

C. Enzymes

Enzymes, thought to be largely of microbial origin (Ladd, 1978), catalyse innumerable reactions in soils and are central to soil biological processes associated with organic matter decomposition and nutrient recycling. They exist in soil in a biotic form associated with viable microorganisms or soil fauna and in forms not associated with living cells, such as excreted enzymes, dead cells, or complexes with organic and mineral colloids (Skujins, 1978; Dick, 1994). The latter type of enzymes may be important in facilitating the hydrolysis of substrates that are too insoluble or too large for microorganisms to use directly (Burns, 1982).

1. Soil enzymes and microbial activity

The activity of about 50 enzymes has been identified in soils (Ladd, 1978). The enzymes most commonly measured are those associated with the C cycle (e.g., amylase, cellulase, glucosidase), the N cycle (e.g., protease, urease), the P cycle (phosphatase), and the S cycle (arylsulfatase). The oxidoreductase enzyme dehydrogenase has also been widely studied, because it has an important role in the oxidation of organic matter (transferring hydrogen from substrates to acceptors), and because it can function only within viable cells. Attempts to correlate enzyme activity with microbial activity in soils have had mixed success, but there is a growing body of evidence to suggest that the activity of certain enzymes is closely linked with microbial activity (Dick, 1994; Nannipieri, 1994). For example, Frankenberger and Dick (1983) demonstrated a strong relationship between the activity of alkaline phosphatase, amidase, and catalase and both microbial respiration and soil biomass, and Asmar et al. (1992) and Fauci and Dick (1994) have shown a strong positive correlation between soil protease activity and both microbial growth and microbial biomass C.

2. Effect of management practices on soil enzymes

Evidence from long-term field experiments suggests that soil enzyme activities reflect different management treatments. Organic amendments such as animal manures, green manures, and crop residues rapidly (within one month) and significantly increased the activity of a range of enzymes compared to unamended soil (Martens et al., 1992). Although these organic amendments also contain enzymes, the increase in activity in soils amended with organic residues likely results from stimulation of microbial activity rather than direct addition of enzymes from the organic sources (ibid.). Soil enzyme activities are also responsive to tillage practices. Gupta and Germida (1988) compared soils cultivated for 69 years with adjacent grassland and found that cultivation depressed phosphatase activity by 49% and arylsulfatase activity by 65%. Other comparisons of tillage treatments have shown that enzyme activity is generally higher in the less-disturbed soil under conservation tillage systems than in soil under conventional tillage systems (Dick, 1994). Martens et al. (1992) have also shown positive correlations between soil enzyme activity and cumulative water infiltration rates.

Pesticides, heavy metals, and other inorganic and organic compounds have significant direct and indirect effects on the activity of various soil enzymes (Schaffer, 1993). The outcomes depend on the complex relationships between the compound and the microbial population, the compound and the enzyme, and the compound and soil colloids, and may be stimulatory or inhibitory (Schaffer, 1993; Nannipieri, 1994).

3. Soil enzymes as indicators of soil quality

From the perspective of soil quality, soil enzyme activity has the potential to be mechanistically related to other soil properties. Based on growing evidence that the activity of certain soil enzymes is correlated with soil biological activity, soil enzyme activity could be used as a sensitive integrative measure of change in the biology and biochemistry of the soil caused by external management or environmental factors (Dick, 1994). The rapid response of some enzymes (e.g., phosphatase, β-glucosidase, urease) to management-induced change in the soil (Martens et al., 1992) could also prove useful and serve as an early warning of impending change in soil quality.

D. Indices of microbial activity

There are various indices of microbial activity. As indicated earlier, the flux of CO_2 or C respiration has traditionally been utilized as an indicator of the metabolic activity of heterotrophic microbes. More recently, the advent of relatively rapid chemical methods to estimate soil microbial biomass has encouraged the use of combinations of attributes to characterize microbial activity. For example, specific respiratory activity (the ratio of CO_2 respired/microbial biomass C) has been used to assess the activity of the microbial biomass. This index has proven useful for explaining differences in microbial biomass caused by different treatments (Anderson and Domsch, 1985) or environmental stress (Brookes, 1995). Dinwoodie and Juma (1988) used the specific respiratory activity to compare carbon cycling in two soils and reported that more C was lost by microbial respiration from the soil with a smaller microbial biomass. Insam et al. (1991) reported a negative correlation between specific respiratory activity and soybean yield at three long-term field experiments. Taken together these data have important implications in agricultural soils: if more C is lost (by respiration) with lower C input, then more care must be taken to maintain organic matter levels (Dinwoodie and Juma, 1988).

V. PLANTS

Plants are generally in direct contact with both the soil and atmosphere, and their responses to that environment (including the presence or absence of specific species or groups, plant visual appearance, growth, reproduction, economic yield, and tissue nutrient content) have been useful indicators of site quality. In this capacity, they have been referred to as biomonitors. Plants, usually agricultural crops, have also been used as indicators of soil quality, or more specifically of soil parameters affecting soil quality. Crop status at some point in the growing season, or crop yield (which integrates conditions over the season), has been related to soil parameters such as fertility and moisture-holding capacity. However, the crop remains part of the soil–plant–atmosphere continuum, making it difficult to isolate the crop response to soil parameters from that to other non-soil influences. Thus, the fact that crop status and yield are measures that integrate the net effect of soil and atmosphere conditions is both the strength and the weakness of using these measures as indicators of soil quality.

A. Plants as indicators of site quality

Plants have been used to characterize the stage of development and stability of so-termed 'natural' ecosystems, such as deserts and wetlands (e.g., Tiner, 1993). They have also been used as sensitive indicators of the presence of contaminants in the air, water, and soil (Meyer et al., 1990). Lichens and mosses have attracted particular attention as biomonitors of air pollution because of their known capacity to accumulate trace metals and organics in their tissues. For example, cadmium was measured in *Sphagnum fiscum* to detect atmospheric deposition of this element across Canada (Glooschenko, 1989). Higher plants may also be suitable for biomonitoring. For example, Gjengedal and Steinnes (1994) showed that *Vaccinium vitis idaea* is a suitable indicator of soil acidification, based on its uptake of rubidium, and also of soil contamination by lead. *Salix phyicifolia* appeared to be a sensitive indicator of soil contamination by cadmium.

Higher plants have been most extensively used as indicators of soil parameters, such as surface soil water pH in organic soils (Swanson and Grigal, 1989), soil salinity (Ataev, 1983), and soil metal content (Davies, 1993; Nashikkar and Chakrabarti, 1994), which affect site quality for agricultural production. Guidelines for using indicator plants to rate site quality, particularly with regard to heavy metal contamination, continue to evolve as knowledge of factors affecting phytotoxicity grows and pressure to apply contaminated material on agricultural land increases. Recently, McBride (1995) questioned the reliance on corn (*Zea mays* L.) as an indicator of heavy metal contamination in sewage sludge applied to agricultural land, as corn is more deeply rooted and more metal-tolerant than other crops. Despite the need to occasionally adjust guidelines, plants are sensitive, integrative indicators of site quality. Not only does the visual appearance of plants and change in species composition indicate the existence of contaminants, but the chemical analysis of plant material can provide a measure of the environmental mobility of a contaminant (Gregson et al., 1994).

B. Crop status and yield as indicators of soil quality

Perhaps the most visible indicator of crop status is plant growth. Plant growth is regulated by a number of soil properties, most notably those that contribute to the soil's ability to provide water, nutrients, and a suitable substrate for seedling emergence and root penetration. Poor vigour and/or stand establishment most frequently result from unfavourable weather conditions and poor soil quality. Crop yield may be viewed as a measurable output from a farming system that integrates the effects of a series of inputs, both climatic (e.g., solar energy and rainfall) and anthropogenic, representing management decisions (e.g., tillage, fertilizer, pesticide, and irrigation water). How one assesses crop yield in terms of efficient use of resources must ultimately include consideration of the longer-term effects the inputs have on soil quality. In the short term, crop status and yield have proved to be useful direct indicators of parameters affecting soil quality.

Within managed agricultural soils, the crop has long been recognized as an indicator of soil fertility, especially N and P. As early as 1910, plant transpiration was considered an indicator of soil fertility (Harris, 1910). Although the emphasis has been on identifying the crop nutrient range required to maximize crop production and seed quality (e.g., Peltonen, 1992), in the last five to 10 years tissue

N concentration has been tested and found to have potential as an indicator of soil N availability, particularly to corn (Fox et al., 1989; Binford et al., 1992). Lopez-Cantarero et al. (1994) found that increases in soil N led to increases in foliar concentrations of chlorophyll a and b, while total chlorophyll concentrations were directly correlated with the level of P in the soil. Crops have also been indicators of available soil water. Various plant responses have been correlated with available soil water, including the visual symptom of leaf rolling (Loresto and Chang, 1981) and early morning leaf water potential measurements (Dwyer and Stewart, 1985). Recent research has suggested that roots in dry soil produce an inhibitor that reduces shoot growth before leaf water status is affected, and that this root-sourced signal is a potential indicator of available soil water (Gowing et al., 1990).

Agricultural research has traditionally focussed on identifying soil parameters and interactions resulting in crop yield variability, often using a regression approach (Bruce et al., 1990) or simulation modelling (Shaffer et al., 1995). Identification of crop responses that are correlated with specific soil parameters reverses the process used in developing these functional relationships among growth, yield, and soil parameters. However, a major benefit of developing and linking functional relationships in a mathematical model of crop yield is the separation and identification of the effects of soil quality, climate and management on yield (Eghball et al., 1995). To date, crop yield response to soil quality cannot be easily separated from yield response to management and climate, although it is generally recognized that stable, economically viable crop yields (not requiring increasing management costs) are associated with good soil quality (e.g., Gregorich and Acton, 1995). One integrated approach to considering yield in the context of soil quality, climate and management has been the development of a framework for evaluating sustainable land management (Smyth and Dumanski, 1995). Crop yield trends are assessed together with indicators of economic viability, environmental impacts, the social context, and level of production risk. In a case study of cereal–livestock operations in Alberta (Gameda and Dumanski, 1995), it was found that extensive regional and farm-specific data are required to quantify the interrelationships among the indicators and to make management recommendations that favour both production and soil quality.

Yield has also been a focus at the production level in recent years as precision farming has provided a means to measure and potentially manage yield variability on a field scale (Franzen and Peck, 1995). At this level, yield variability is most frequently due to spatial variability in topography (Simmons et al., 1989), which affects soil parameters such as drainage, texture, and organic matter content, which in turn determine water and nutrient-holding capacity, root penetration, and rooting volume. Crop yield is the primary indicator of spatial variability in soil parameters. However, in-season indicators of crop status may be more directly related to soil parameters affecting soil quality, because they do not integrate over the whole season as yield does and so may be less influenced by weather variability.

Several measurements have been developed to monitor crop status. Chlorophyll meter readings have been used to assess leaf greenness and leaf N concentration (Schepers et al., 1992; Dwyer et al., 1995). Remote sensing techniques, developed for monitoring crop growth conditions over large areas, have provided useful relationships for evaluating crop status at a field level. Light reflectance is negatively related to leaf chlorophyll content at wavelengths near 550 nm (Blackmer et al., 1994) or 600 nm (Adcock et al., 1990) and positively related to green leaf area at wavelengths near 800 nm (Adcock et al., 1990). A normalized difference vegetation index (NDVI) or the canopy light reflectance at (800 nm−600 nm)/(800 nm + 600 nm), integrates field greenness, and field leaf area and has been found to be closely related to plant above-ground dry matter (Rouse, 1974). Ma et al. (1996) demonstrated that both canopy light reflectance and field greenness measured at preanthesis in corn were correlated with grain yield at harvest. Canopy reflectance measured after anthesis differentiated hybrid differences in leaf senescence. These in-season indicators of crop status are adding to baseline data used to explain yield variability and may also be interpreted as indicators of threshold values for soil quality parameters limiting yield, such as soil pH, water-holding capacity, and fertility. As experience and proficiency are gained in interpreting yield variability on a spatial grid, soil and crop management recommendations are aimed at maximizing yield across the field in the short term, while maintaining or improving soil quality in the longer term.

Relationships between crop status or yield and soil quality are also becoming more evident. In agricultural production systems dependent on high input management, the relationship between yield or crop status indicators and soil quality parameters is masked, and in extreme situations the soil is reduced to a substrate to anchor crop roots. However, as overall efficiency of production is increasingly important to the economic viability of production, anthropogenic inputs are minimized, and the positive correlation between stable yields over time and soil quality is emerging.

VI. CONCLUDING REMARKS

Some biological attributes discussed in this chapter are sub-attributes or subsets of other attributes (Gregorich et al., 1994). Light fraction, microbial biomass, and mineralizable organic matter are labile, dynamic forms of soil organic matter, itself an indicator of soil quality. Table 4.3 summarizes the main soil biological parameters discussed in this chapter, along with their specific potential as an attribute of soil biological quality. In conclusion, the following factors affecting the use of biological attributes should be considered.

- In most cases, biological attributes should be expressed on a soil volume basis, rather than a concentration basis, to accommodate changes in soil density (Doran and Parkin, 1994).
- Cropping history and seasonal variability affect biological attributes and must be taken into account when comparing soils.
- Usually it is assumed that higher levels of these attributes represent improvements in soil quality. However, care is needed when evaluating these dynamic indicators over the longer term. Reliability depends on the mechanism of dynamic soil organic matter accumulation (Janzen et al., 1997).

TABLE 4.3

Selected soil biological attributes and some comments on their application

Attribute	Comments
Total organic C and N	A coarse measure of soil quality. Increasing levels associated with available plant nutrients and good soil structure
Light fraction	A labile organic matter fraction responsive to soil management, and generally rapidly reduced under conditions of low C inputs to the soil
Microbial biomass	Responds rapidly to changes in soil processes as a result of management. The ratio of biomass C to total soil C can provide an index of organic matter dynamics
Fauna	Sensitive to soil pollution and degradation. Play an important role in soil processes, such as development of biopores and mixing of organic and mineral components
Mineralizable C and N	Reflects microbial activity and levels of labile C in soil. Mineralizable N integrates capacity or potential of soil to supply mobile forms of N
Enzymes	Provide a sensitive measure of changes in microbial and biochemical activity in a soil
Plants	Integrate many natural and anthropogenic soil inputs, and reflect spatial change and variability in soil properties and quality

- The sampling protocol in general, and the time of sampling in particular, need to be carefully considered in order to avoid temporal variability, which may obscure real differences between attributes.
- Limiting levels, values, or thresholds for biological properties in soils are generally not known (Sparling, 1997). As yet, the absolute value of any single biological attribute is of limited use for assessing soil quality for crop production. Therefore it is important that some comparison to a reference or baseline be made when evaluating biological attributes of soil quality.
- Because of their transient nature, the rate of change in attributes, rather than the absolute value, may provide a better assessment of soil quality over the long term.
- Due to the multifunctional nature of soil organic matter in soil quality and ecosystem health, use of a suite of biological attributes is generally required to provide a comprehensive view for different functions, such as soil structure, nutrient turnover and storage, and biological activity.

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Chapter 5

AN ECOSYSTEM PERSPECTIVE OF SOIL QUALITY

B.H. ELLERT, M.J. CLAPPERTON and D.W. ANDERSON

I. INTRODUCTION

Soils are critical components of terrestrial ecosystems, which also include the atmosphere, water, plants, and other organisms. Healthy or good quality soils are essential for ecosystems to remain intact or recover from disturbances, such as drought, climate change, pest infestation, pollution, and human exploitation, including agriculture. Most concepts of soil quality have been based on the premise that various soil components influence the capacity of soils to fulfill specified functions. These functions are interrelated and often include support of plant growth (e.g., regulation of water and nutrient availability), environmental buffering (e.g., regulation of element fluxes and of air and water composition), and serving as a reservoir of genetic information (especially microbes and seeds).

A. The big picture

Soil, or the *pedosphere*, is an intersection among the lithosphere, hydrosphere, atmosphere, biosphere, and noösphere (Fig. 5.1). Vernadsky (1945) recognized humans as a major geological force, and adopted the term *noösphere* to account for the influence of the human mind on earth processes. This influence is particularly conspicuous in agricultural systems. The ecosphere, as coined by Cole (1958), is distinct from the biosphere alone (i.e., the collective sum of all living matter) and encompasses interactions among all five major spheres (i.e., litho-, hydro-, atmo-, bio- and noö-spheres). An ecosystem perspective treats soils as essential but insufficient components of the ecosphere.

An ecosystem perspective of soil quality emphasizes that soils and soil functions influence and are influenced by several other ecosystem components and functions (Fig. 5.1). This perspective emphasizes that soils are nested within larger ecosystems, whereas most concepts of soil quality emphasize smaller, almost dimensionless or point attributes of soils (e.g., plant-available nutrients). This chapter considers how ecosystem components interact to develop and support productive soils.

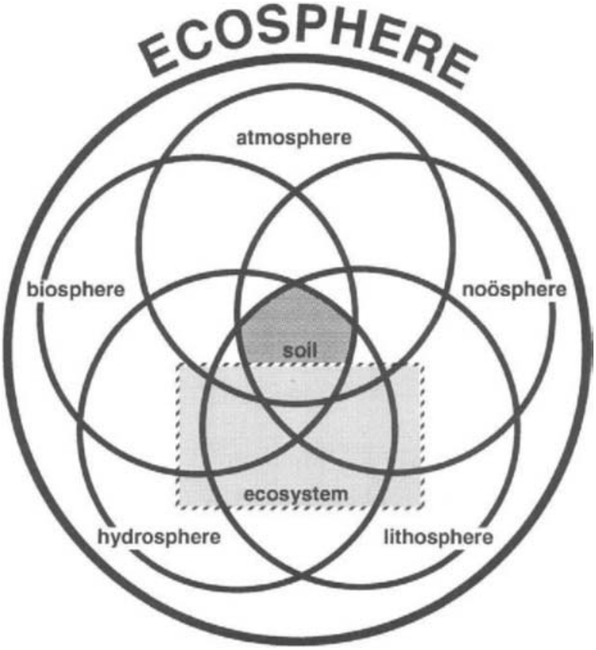

Fig. 5.1. Illustration of the ecosphere encompassing the five major spheres and various interactions, and of soils or the pedosphere as the interface among the major spheres. Arbitrary ecosystem boundaries encompass a sub-set of the ecosphere.

B. Descriptive versus prescriptive approaches

Pedologists, those scientists most concerned with soils as natural occurrences in a field setting, along with ecologists, naturalists, and geographers (to name a few), have long perceived soil as an ecosystem component. Their work in the field deals with the nature and classification of soil profiles in relation to parent material, vegetation, and microclimate as influenced by topography. Their knowledge tends to be more descriptive, intuitive (governed by experience), and conceptual rather than quantitative (Hudson, 1992).

Agronomists, on the other hand, are concerned primarily with the production of food and fiber, and perceive soils mainly as media to support plant growth. Fertility trials, crop rotation studies, and tillage experiments have shown how these media may be manipulated to achieve specific, production-related goals (Nikiforoff, 1959). As a quantitative science, founded on objective testing of rival management practices, agronomy has helped to increase or maintain food and fiber production. Despite these successes, agronomic prescriptions also have had undesirable side-effects, such as nutrient depletion; erosion; compaction; and pollution of soil, water, and air. Many of the problems may not result from deficiencies in the research, but from inappropriate application of practices on large scales or to poorly suited soils. For example, monocultures may be productive in the short term, but the build-up of pests often impairs long-term viability. Management practices that work well on uniform soils often are inappropriate for farm scales that encompass the variations typical of many soil landscapes.

The objective of this chapter is to explore how an ecosystem perspective may contribute to an understanding of soil quality. By ecosystem perspective, we mean one that emphasizes ecosystem structure and functioning at large scales in space and time, rather that at single points. By combining the broad-scale, conceptual approaches of field ecologists and pedologists with the quantitative and prescriptive approaches of agronomists, we hope to benefit from the best of both approaches. Jackson and Piper (1989) endorsed a marriage between ecology and agriculture. Extensive studies from an ecosystem perspective are helping to identify and lessen the undesirable side-effects of traditional soil management practices (Oberle and Keeney, 1991; Peterson et al., 1993). Recent innovations, such as landscape-scale agronomy (Pennock et al., 1994) and precision agriculture (Vanden Heuvel, 1996) hold promise, and are likely to require both pedological, ecosystem-based knowledge, as well as quantitative agronomic studies.

II. SOILS AS COMPONENTS OF ECOSYSTEMS

Students of soil science often are instructed that five or more "soil-forming factors" influence soil development: soils = *f*(climate, organisms, topography, parent material, time ...). Original interpretations of these factors implied a linear "cause-and-effect" relationship between soil formation and each factor. However, because the factors were interdependent, circular arguments often arose about whether specific factors were "causes" or "effects". Consequently, Jenny (1961) revised earlier interpretations when he wrote, "*Moreover, the factors are not formers, or creators, or forces; they are variables (state factors) that define the state of the soil system.*" He emphasized that ecosystem components or properties, such as soils, vegetation, and animals were functions of three state factors: ecosystem property = f(initial state, external pressures or potentials, time) (Fig. 5.2). Fluxes or exchanges of matter and energy between the system and its surroundings were governed by gradients between the outside and inside of the system.

Thus Jenny clarified what the earlier pedologists and naturalists had recognized: soil cannot be considered in isolation from the rest of the ecosystem. For example, even though climate and parent material often are regarded as external pressures, it is now clear that biota profoundly influenced atmospheric composition and climate and reworked the earth's external crust throughout geological history (Gorham, 1991; Lovelock, 1989). In an analysis of interactions among biota and soils, van Breemen (1993) concluded that "*soils derive their existence from life processes on a global scale, and are made more fit for plant growth by life processes on the ecosystem scale.*" The functional relationship between ecosystem properties and the state factors has proved difficult to evaluate, because environmental variables function over multiple time scales, and the factors are difficult to combine in a single model (Phillips, 1989).

The external pressures that shape ecosystem development depend on the boundaries or scales (in time and space) of the ecosystem (Fig. 5.2). For example, atmospheric composition may be a static variable on the time scale of a single decade, but changes during industrialization have had significant impacts on the scale of a century or two. In addition to the biophysical boundaries depicted in

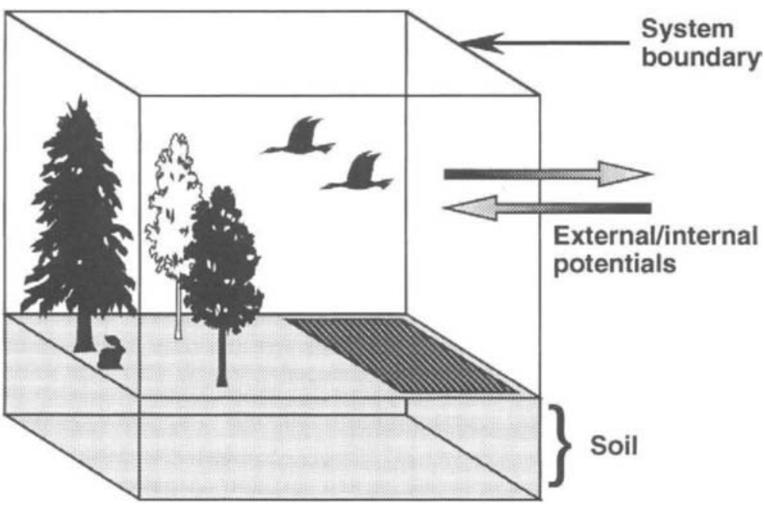

Fig. 5.2. An ecosystem perspective recognizes soils as a component of a system that is defined by initial conditions or states within the system boundary, changes through time and exchanges of matter and energy between the system and its surroundings.

Fig. 5.2, agroecosystems sometimes are viewed as ecosystems nested within socio-economic boundaries (Conway, 1987). The socioeconomic dimensions of agroecosystems recognize interactions among humans, their institutions (including communities, marketing systems, agri-business, governments, religion), and natural ecosystems (Berry, 1991). Government policy that influences land use, for example, has caused both intentional and unforeseen changes in soil carbon storage and erosion on agricultural land (Gebhart et al., 1994; Rosaasen and Lokken, 1994). While agronomists are accustomed to working within socio-economic constraints, ecologists increasingly are being asked to consider the impacts of manipulation to meet human goals as the field of "ecosystem management" emerges (Slocombe, 1993; Underwood, 1995; Ecological Society of America, 1995).

III. SPATIAL AND TEMPORAL DIMENSIONS

Ecosystem processes and components vary highly in time and space. Organization of this complexity according to the temporal and spatial scales of processes and components has proved invaluable to understanding ecosystem structure and function (di Castri and Hadley, 1988). Soil surveyors have traditionally dealt with spatial variability by developing conceptual models of soil landscapes, which are continuously tested and modified by examining pedons during the course of the survey (Hudson, 1992). Soil quality has been investigated at spatial scales ranging from mm (e.g., rhizosphere effects) to thousands of km (e.g., soil variation among major biomes), and at temporal scales from seconds (e.g., ion exchange) to centuries and epochs (e.g., pedogenesis) (Fig. 5.3). Some processes (e.g., sorption of ortho P on colloidal surfaces) tend to occur at much smaller temporal–spatial scales than others (e.g., pedogenesis on glacial deposits).

A grouping of soil processes based on time was given by Stewart et al. (1990), who distinguished between highly dynamic (e.g., translocation of soluble nutrients), dynamic (e.g., sorption and desorption nutrients), and more static (e.g., release of nutrients during mineral weathering) processes. As with any generalization, the relationships in Fig. 5.3 are not to be construed as smooth regular transitions among scales; many ecosystem components are patchy (e.g., slope positions in hummocky terrain) and subject to threshold processes (e.g., floodplain formation) (Muhs, 1984). Episodic events, such as an intense rain storm that is an order of magnitude greater than the average can result in marked changes in soils (Stewart et al., 1990).

A. Interactions among multiple scales in time or space

An ecosystem perspective considers interactions among processes at multiple scales in time or space, whereas agronomic studies usually focus on single scales. Ecosystem processes at a particular scale are constrained by driving variables at higher levels and explained by mechanisms operating at smaller scales (O'Neill, 1988). Rhizosphere processes are constrained by the characteristics of the plant root system, which in turn is constrained by the depth of soil development (Fig. 5.3). The significance of solutes and solute exchange with colloidal surfaces, and of rhizosphere

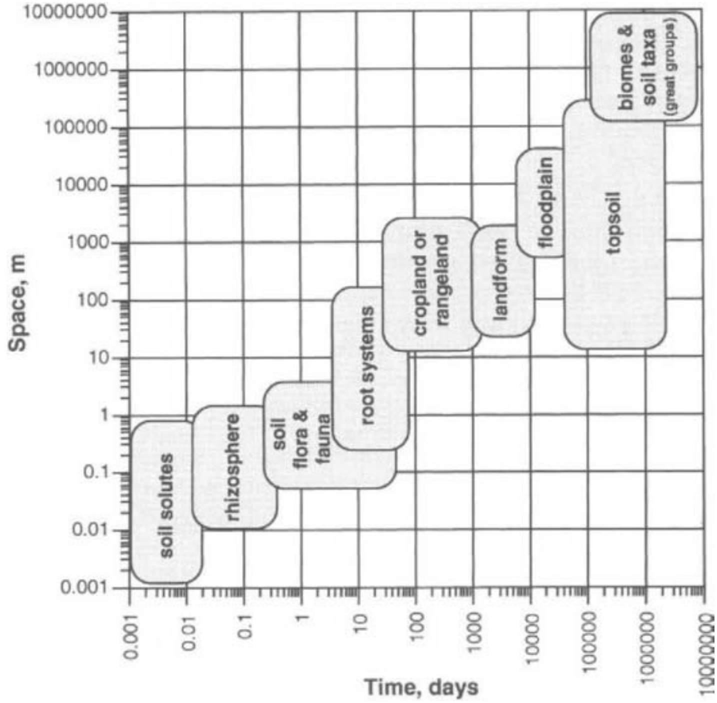

Fig. 5.3. Temporal and spatial scales of ecosystem components (and associated processes) that directly influence soil quality.

processes becomes evident at larger scales, such as productivity of cropland and rangeland or even pedogenesis.

Considerable advances in soil management have been achieved through studies at small scales (<1000 days and m). Information at larger scales has been obtained from studies of soil genesis, long-term crop rotations, and forest watersheds, and more recently, of remotely sensed landscape data (Anderson, 1991; Jenkinson, 1991; Coleman et al., 1992; Burke and Lauenroth, 1993). By exposing trends that were indistinguishable from short-term noise, long-term ecological research is clarifying the interactions between management on the scale of seasons and soil quality on the scale of decades or more (Janzen, 1995). Nested databases, stratified according to temporal and spatial hierarchies, help to expose interactions among scales and ensure that models and other analyses are implemented at the appropriate scale and have adequate linkages to other scales (Dumanski et al., 1993).

B. Scaling up to assess sustainability

The challenge to integrate and interpret information obtained at multiple scales in time and space must be tackled to attain an ecosystem perspective of soil quality. Down-scaling is the reductionist approach whereby an empirically defined system is dismantled to explain how it works according to its constituent processes. Up-scaling, whereby information at smaller scales is used to derive processes at larger scales, is especially challenging, because new interactions and processes often lead to the emergence of properties that were unsuspected from information obtained at smaller scales. This surprise emergence of unforeseen properties concurs with the maxim that the whole is more than a sum of its parts. Up-scaling is imperative to assess ecosystem sustainability, that is, to project how ecosystems might function in the future (large temporal scale) and how various ecosystems might interact (large spatial scale; global environmental change) (Jarvis, 1995).

Soil quality in agricultural land is concerned with maintaining food and fiber production over large areas of space for an extended period of time, hopefully indefinitely. From an ecosystem perspective, soils acquire additional functions, such as the partitioning of water into runoff, stored soil moisture, and groundwater recharge, and serving as an environmental filter or buffer to cleanse and regulate the composition of air and water. Large-scale investigations and methods to scale up information from fragmented studies at smaller scales are needed to improve our understanding of previous changes in soil quality, and to anticipate scenarios that are likely to occur in the future. Organization of soil components according to spatial scale has helped to extrapolate information gained from process studies to larger regions (Anderson et al., 1983; Roberts et al., 1989). Long-term agricultural plots are useful to evaluate the influence of variables that change on the same scale (e.g., atmospheric composition) and of catastrophic events with long (>20 yr) return times, including droughts, pest infestations, and dust storms.

Clarification of the interrelationships among components and processes at different scales is required to help explain ecosystem phenomena. The spatial scale of soils is comparable to that of the atmosphere, but atmospheric processes fluctuate on much shorter time scales than soil genesis. Although some processes within soils operate on small time scales (< 1000 d), many soils appear to be resistant or resilient to fundamental change compared to the atmosphere, vegetation, lakes, and streams. That is, by buffering resource availability (including water, nutrients, and contaminants) and serving as a reservoir genetic information (including seeds, spores, and cysts), healthy soils tend to resist or recover from changes. Interactions among components and processes at multiple scales generate self-organization within ecosystems that resist perturbations, provided the bounds of adaptability are not exceeded (Perry, 1995).

IV. INTERACTIONS AMONG SOILS AND OTHER ECOSYSTEM COMPONENTS

Soils interact with and are influenced by other ecosystem components, including the atmosphere, water, and organisms. When eroding landscapes are revegetated, the soil is anchored, organic matter accumulates, and water runoff is reduced in favor of infiltration and transpiration. All of these processes feed back positively to further enhance revegetation and reduce erosion. Interactions among soils and other ecosystem components were recognized by early scientists, especially those at Soviet institutions that fostered interdisciplinary work among geologists, botanists, and geographers (Sukachev, 1960; Tandarich and Sprecher, 1994). Writing on plant geography in 1807, Alexander von Humboldt stated, "*In the great chain of causes and effects no thing and no activity should be regarded in isolation*" (Major, 1969). Vernadsky (1945), often acclaimed as the founder of biogeochemistry, recognized that the inextricable interactions among living matter (including humans), the atmosphere, and the earth's crust are manifested as nutrient or biogeochemical cycles.

Contemporary ecology has embraced the concept of ecosystems as units of biological organization consisting of communities of organisms interacting with their physical environment. The interdependence among organisms and their physical environment is perhaps nowhere as apparent as in soils, where strong interactions among plants, microorganisms, and soil constituents dictate environmental conditions and species composition. The ecosystem concept recognizes that individual organisms cannot be considered in isolation from other organisms or their physical environment (Odum, 1969).

A. Ecosystems as self-organized, cybernetic systems

The application of systems engineering to describe the structure, function, and dynamics of ecosystems culminated in debates whether ecosystems were cybernetic, that is, whether ecosystems exchanged information among system components and were self-regulating like electro-mechanical systems (Patten and Odum, 1981). Non-cybernetic systems are characterized by linear relationships between cause and effect; stimuli are independent of responses (Fig. 5.4a). Ecosystems, however, are rich in "information" consisting of interactions among system components. Information exchange in cybernetic systems produces non-linear relationships between information inputs (causes or stimuli) and system outputs (effects or responses), because inputs are at least partially determined by outputs. Cybernetic systems are subject to regulation by feedback or a "circular" sequence of events in which a change in one system component triggers stimuli and responses throughout the system that, in turn, eventually alter the original component (Berryman, 1989).

Positive feedback occurs when stimuli and responses tend to reinforce the original change. Negative feedback occurs when the stimuli and responses tend to cancel the original change. Both kinds of interactions are essential: negative feedback promotes homeostasis or stability, whereas positive feedback loops produce evolutionary changes that shift system equilibria or goals. Accumulation of soil organic matter often reflects positive feedback, whereby residues from colonizing plants become mixed with soil parent materials and further enhance plant growth, which again increases organic matter deposition. Later, negative feedback might restrict plant growth when nutrients (e.g., N, P, S) become tied up in organic matter with subsequent nutrient availability regulated by organic matter decomposition.

We suggest that both natural and agricultural ecosystems are cybernetic, but that they differ in the extent to which controllers are concentrated in specific structures. The cybernetic attributes of natural ecosystems emerge from the organization or structure of large, complex, and decentralized systems, with self-organizing feed-

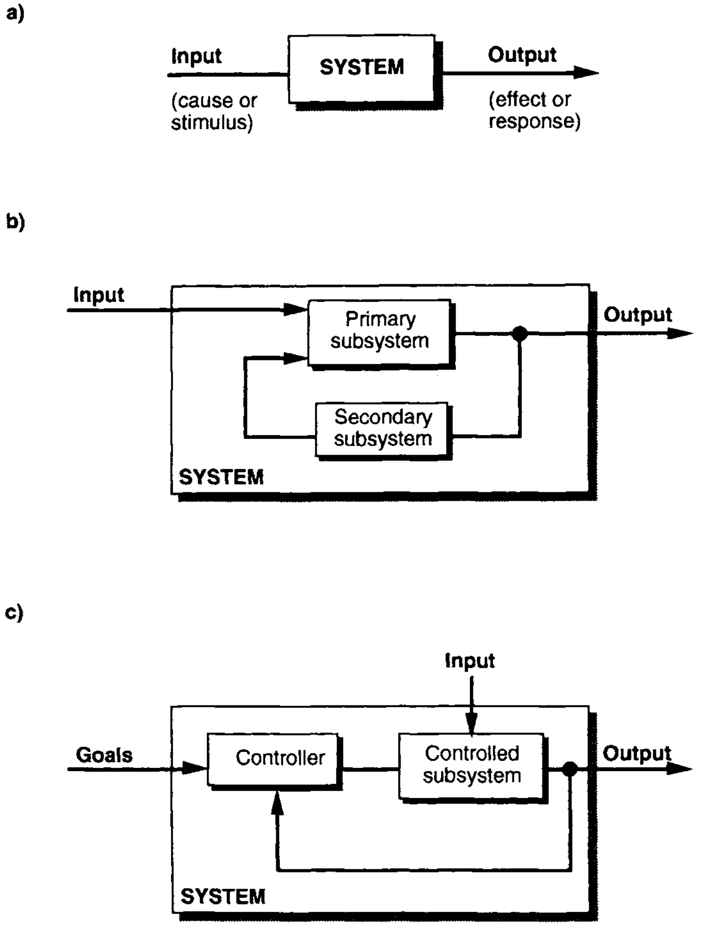

Fig. 5.4. Stimuli and response in contrasting systems: a) in non-cybernetic systems there is a linear relationship between cause and effect; b) in natural ecosystems the response to stimuli are governed by information exchange among several system components without specific goals or controllers; c) humans impose external goals or inputs in an attempt to regulate agroecosystems (modified after Patten and Odum, 1981).

backs generated by the information links among material cycles and energy flows without the involvement of specific goals or controllers (Fig. 5.4b). Self-organization based on information has been described as phenomenological rather than teleological or controlled in a top-down manner by some unknown force. In contrast, humans, to varying extents, seize the controls in agricultural ecosystems, and the structure and function of the system is directed as much as possible to achieve specific goals (Fig. 5.4c). Agricultural ecosystems are more machine-like, because the over-riding goal is to maximize the production of harvestable food and fiber, and humans are the controllers.

B. Human controllers

The effectiveness of human controllers of agricultural ecosystems is open for debate (Jackson and Piper, 1989). The debate is an important one, if the flow of food and fiber from a finite agricultural land base is expected to sustain our massive human population. The continued growth or maintenance of the current human population, however, might be questioned as a desirable goal for the future. But regardless of humanity's goals, agriculture is a human enterprise. Thus, the integrity or health of agroecosystems, including soil quality, depends on the effectiveness of human controllers.

Ecosystem management has been defined as "*management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and processes necessary to sustain ecosystem composition, structure, and function*" (Ecological Society of America, 1995). The concept has raised both technical concerns whether humans have the insight required to manage ecosystems and ethical concerns whether humans have the right to manage ecosystems. Technical concerns are based on our grossly inadequate understanding of ecosystems, as suggested by pest outbreaks in agricultural crops, failure of reforestation efforts, or collapse of the Atlantic cod fishery. Ethical concerns are often based on the anthropocentric bias of ecosystem management. Some suggest that environmental problems might best be confronted as fundamental challenges to human values rather than technical challenges to be managed (Jamieson, 1992). Perhaps an ecosystem perspective might provide the insight required to confront challenges both to manage ecosystems and to reassess values pertaining to ecosystems.

C. Ecosystem management based on an understanding of interactions

Effective management or control of agroecosystems and soil quality requires an understanding of the interactions or information linkages among system components. Despite impressive advances in agronomic production, an ecosystem perspective of soil quality indicates that current understanding of interactions among plants, microorganisms, and soil constituents is, at best, rudimentary. For example, cereal breeders tend to "short-circuit" information linkages by selecting for plants that are less dependent on mycorrhizal symbioses (Hetrick et al., 1993). Information linkages include both competitive and mutualistic interactions among organisms, as well as interactions among organisms and their physical environment.

Darwinian evolution stresses the importance of competition among species, but mutualism is equally important in many systems. Kropotkin, a Russian familiar with the sparse biota and harsh climate in Siberia and Manchuria, observed that the ability of an organism to adapt to or modify its environment was equally important to its success in competitive struggles, and that struggles sometimes lead to cooperation (Gould, 1988). Mutualistic interactions between fungi and plant roots strongly influence plant succession and soil genesis, because mycorrhizae improve plant nutrition (especially P) and soil structure (Reeves and Redente, 1991; Perry et al., 1990). Colonization of root systems by mycorrhizal fungi provide a competitive advantage to the resulting mycorrhizae, allowing proliferation of the root system and deposition of soil organic matter that enhances soil structure and microbial habitat.

Interactions between plants and decomposer organisms in the soil are crucial to maintaining nutrient cycling in natural and agricultural systems: plants deposit energy-rich substrates in the soils, and nutrients are released by the activity of decomposers in the soil. Soil microbes responsible for nutrient mobilization assimilate the nutrients required to maximize their own biomass (under the constraint of decomposable organic matter availability), and release the remaining nutrients for plant uptake (Harte and Kinzig, 1993). Soil micro- and meso-fauna (i.e., consumers in detrital food webs) have a much greater impact on soil nutrient dynamics than expected from their small biomass relative to the microflora (Hunt et al., 1987). Recent data suggest that the structural chemistry of soil organic matter is determined primarily by interactions between decomposer organisms and soil minerals, rather than by the chemical nature of the original plant residue inputs (Randall et al., 1995). Although it is accepted that soil organisms and plants interact through the production of chemical signals (e.g., hormones, carbohydrates, nucleic acids, chelates, proteins) (Halverson and Stacey, 1986), other data suggest that soil physical conditions may also induce hormonal signals in plants (Passioura, 1991).

Integrated pest management uses information on interactions among plants (crops and other hosts), pests (often fungi or insects), and the environment (soil conditions, aerial climate) to prevent excessive losses to pests, whereas conventional pest control methods focus on pest eradication without considering ecological interactions (Edens and Haynes, 1982; Altieri, 1991). Recent studies suggest that mycorrhizal fungi influence competition between crops and weeds (Clapperton and Blackshaw, unpublished). A greater understanding of ecological interactions is required to discover what information linkages have been severed in simplified agroecosystems, and what linkages might be established or manipulated to achieve management goals.

V. ECOSYSTEM STABILITY, DIVERSITY, AND COMPLEXITY

Relationships among structural complexity, biodiversity, and ecosystem stability arise from interactions or "information exchanges" among system components. Development of stable climax communities traditionally was regarded as the pinnacle of ecological succession (Odum, 1969). During succession, interactions among ecosystem components forged linkages that favored homeostasis (i.e., coordinated responses among components compensated for perturbations). Later it was recognized that although stable systems did resist perturbations, they also returned to the original states when displaced from equilibrium (Holling, 1986). Resistance refers to the extent of disturbance before the ecosystem is displaced from equilibrium; resilience refers to the ability and time required for the ecosystem to return to its original equilibrium state after being perturbed.

A. Ecosystem equilibria and dynamics

Succession accounts for ecosystem dynamics as the interplay between exploitation and colonization of disturbed land, followed by conservation of energy and nutrients to stabilize the community at equilibrium. In addition to exploitation and conservation phases of ecosystem change, Holling (1986) suggested that ecosystems also undergo "creative destruction" and reorganization. He suggested that increasingly strong connections and interactions in mature ecosystems eventually lead to an abrupt change or surprise. The abrupt change (release of stored capital resulting in creative destruction) may then trigger reorganization and renewal of the ecosystem as large stores of soil organic matter and nutrients again are available for exploitation. Thus, systems with many connections and strong interactions become brittle or fragile, and the probability increases that the system will change to some new, rather than the original, equilibrium state. The impact of the abrupt change is strongly determined by soil nutrient dynamics, because total collapse is possible when nutrient mobilization is not balanced by retention. The potential for abrupt change suggests that the ability of agroecosystems to recover from change and undergo renewal may be equally or more important than long-term stability.

While the information networks and feedback loops in ecosystems allow for self-organization and dynamic equilibria, they also restrict the insight gained from reductionist studies of individual processes. Such studies in simplified agroecosystems have contributed substantially to food production, but the behavior of more complex systems is very difficult to infer from studies of individual processes. Feedback or information linkages in more complex, cybernetic systems introduce non-linearity such that stimuli are modified by responses. This non-linearity, in systems conforming to deterministic rules, may culminate in fundamental randomness or "chaos" that makes system behavior extremely sensitive to initial states and practically unpredictable. If atmospheric dynamics are subject to deterministic chaos, predictability of the weather will be severely impaired, but weather conditions will be confined within specific limits (Tsonis, 1989). Although the potential for deterministic chaos exists in most ecosystems, Berryman and Millstein (1989) suggested that natural ecosystems do not normally behave chaotically, but that human intervention may push ecosystems (especially agricultural ones) into the chaotic regime.

B. Relationship between complexity and stability

Understanding the implications of unprecedented rates of species extinction for ecosystem function requires integration of information on population ecology and systems ecology (Schultze and Mooney, 1993). Population ecology traditionally has focused on interactions within and among individual species, whereas systems ecology emphasizes interactions (especially energy and nutrient flows) among functional groups of organisms and their environment. Initial work on ecosystem dynamics emphasized the equilibrium view, in which increasing complexity or diversity (more species, trophic levels) enhanced stability. Conventional wisdom ("don't put all your eggs in one basket") suggested that more pathways of energy and information transfer enhanced stability, and as a consequence natural areas with several species were thought to be more stable than monoculture cropland (Pimm, 1984). Theoretical work, however, suggested the opposite: complex systems with longer food chains were less stable or resilient, and were more brittle.

Ongoing work on ecosystem complexity, biodiversity, stability, and resilience will be crucial to understanding the sustainability of ecosystems and soil quality. Microcosm studies suggested that plant productivity was greater in systems with higher diversity (Naeem et al., 1995). Field data indicated that diverse grasslands in Minnesota, U.S.A., were more resistant to, and recovered more fully from, a severe drought in 1988 compared to grass swards with four or fewer species (Tilman and Downing, 1994). In a subsequent study, direct manipulation of the number of grass species (but not the particular combination of species) clearly indicated that grassland productivity and N utilization and retention increased with plant diversity (Tilman et al., 1996).

Other studies indicate that the species composition of plant communities is strongly dependent on that of soil organism communities (Bever, 1994; Johnson et al., 1992). Since the diversity of organisms within topsoil is very high relative to that of above-ground organisms, sometimes it is assumed that many species of soil organisms fulfill the same function and thus are redundant. Currently, however, we are unable to predict which species (few soil organisms have been classified at the species level) may be eliminated without adverse consequences. Beare et al. (1995) suggest that complex interactions among a myriad of soil species at multiple scales may be required to regulate biogeochemical cycling.

C. Complexity of agroecosystems

The complexity of agroecosystems is a function of the number of species (including crops, livestock, pests, and shelterbelt trees) as well as the spatial and temporal dimensions of the systems (Fig. 5.5). The importance of biodiversity for the functioning of agroecosystems often is discussed (e.g., the Irish potato famine of 1845 is attributed to over-reliance on a single crop) but remains uncertain because data are scarce and analytical approaches are rudimentary. Monoculture provides financial and organizational benefits but also increases economic risk and environmental degradation (including soil erosion, water pollution, and fossil fuel depletion) (Power and Follett, 1987).

In North America, cereal monocultures occupy vast homogeneous tracts of land (low spatial complexity) and are controlled using similar management practices year after year (low temporal complexity). The increased use of crop rotations adds complexity, because two or more species are used, the area occupied by each crop declines, and management practices recur with a frequency of two or more years. Integrating crop and livestock production on a single farm or locale provides additional opportunities to enhance biodiversity and promote nutrient recycling (Fig. 5.5). In developing countries, shifting cultivation of several crops (e.g., cassava, beans, peanuts, rice, and maize) occurs on a patch-work mosaic of plots that are

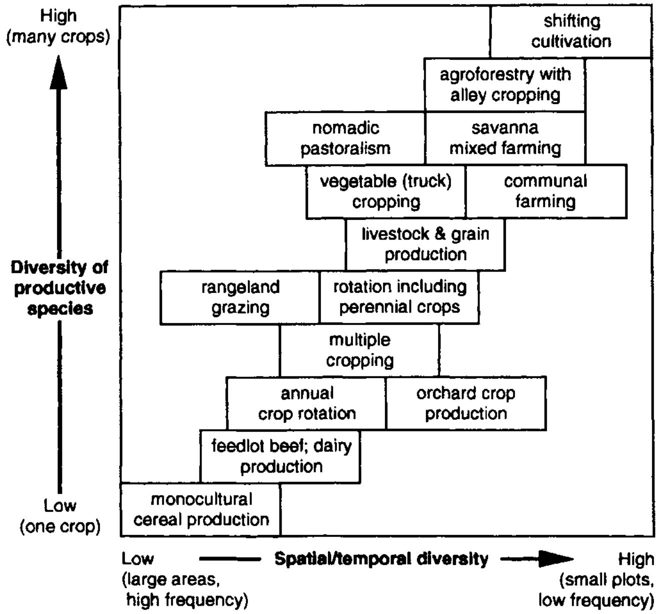

Fig. 5.5. Classification of agricultural systems on the basis of biodiversity of crop and livestock species and spatial/temporal complexity. The horizontal axis represents both the spatial complexity (ranging from large homogeneous fields to small patch-work plots) and the temporal complexity (with management practices and crop development repeated with frequencies ranging from one year in annual monocultures to several years in complex crop rotations and shifting cultivation) (modified after Swift and Anderson, 1993).

used for two to five years before the plots are abandoned for several years while the soil rejuvenates (high spatial and temporal complexity) (Stewart and Tiessen, 1990). Diversity may not necessarily confer stability on agroecosystems. For example, the potential for destabilizing pest infestations may increase if a host plant is included year after year (Loomis and Connor, 1992).

Agronomic management has a marked influence on the diversity of soil organisms. The decomposition sub-system under no-tillage at the Horseshoe Bend site in Georgia, U.S.A., was dominated by fungi and earthworms, whereas that under conventional tillage was dominated by bacteria and nematodes (Hendrix et al., 1986). Perry et al., (1990) observed that maintenance of biodiversity is crucial to ensure that linkages among plants, soil organisms, and soil minerals are not severed. They suggested that reforestation failures in some areas of Oregon and California may have been caused by the eradication of mycorrhizal fungi, which are required to link trees and soil. Wild flora and fauna often are displaced from agroecosystems by physical (e.g., tillage), chemical (e.g., pesticides), and biological (e.g., plant cover) disturbance, but various management options (e.g., crop rotation, intercropping, and conservation tillage) are available to help preserve wildlife habitat in agricultural landscapes (McLaughlin and Mineau, 1995).

VI. INDICATORS OF SOIL QUALITY AND ECOSYSTEM HEALTH

An ecosystem perspective emphasizes the fascinatingly complex linkages among soils and other ecosystem components and processes. The challenge to assess the influence of humans on these components and processes has culminated in attempts to measure soil quality or ecosystem health. Measurement of soil quality and ecosystem health remains elusive, because our understanding of ecosystem structure, function, and dynamics is rudimentary. It is clear, however, that soil quality and ecosystem health are closely linked (Fig. 5.6). The precise definitions of both soil quality and ecosystem health are controversial, but the concepts about the relative fitness of soils and ecosystems to serve various functions are likely as old as agriculture.

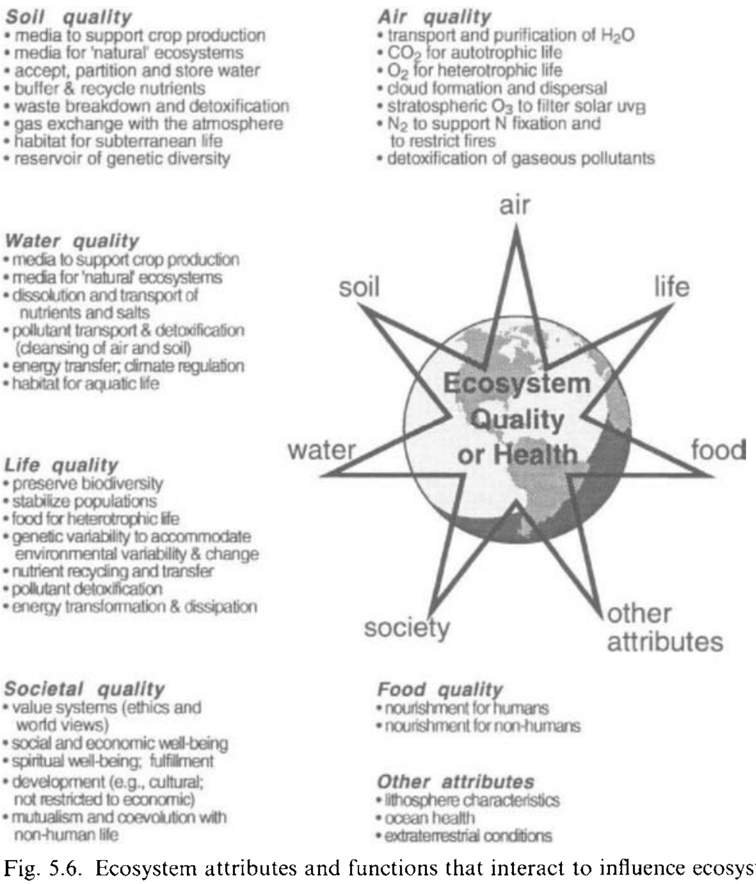

Fig. 5.6. Ecosystem attributes and functions that interact to influence ecosystem quality or health. From an ecosystem perspective, soil quality is one attribute of ecosystem health, crop production is one function defining soil quality, and both influence and are influenced by other attributes and functions, such as water quality and preservation of biodiversity.

A. Quality defined as the capacity to provide services

As a prerequisite to measuring soil or ecosystem quality, the functions or services expected of the system must be delineated. In addition to the economic and sustainable production of food and fiber, an increasing array of services is expected from agroecosystems, including water and gas partitioning, wildlife habitat, environmental filtering, waste disposal, biodiversity reservoir, and aesthetic retreat (Fig. 5.6). Some of the services may be compatible, and others may require trade-offs. A unique set of conditions may be required for each service or even each crop (e.g., blueberries grow best on acidic soils, whereas alfalfa requires neutral soils). Since a wide range of ecosystem and soil attributes likely will be required to assess fitness for production of various crops, coarse or general indices with wider applicability may be most useful. An ecosystem perspective focuses on structures, functions, and services as influenced by soil, as well as air, water, life or organisms, society, and other attributes (Fig. 5.6).

In this volume, soil quality is defined as the fitness of a soil for crop production. Extending this definition to the ecosystem level, agroecosystem quality or health refers to the fitness of the crop production system. It is important to recognize that the concepts of soil quality and ecosystem health are subjective and human-designed (anthropocentric); quality refers to the ability of the soil or ecosystem to perform a defined function, and the functions are selected by humans (i.e., they represent human ideology or values). Frank discussions about the values involved in such concepts may be equally or more important than the technical development and use of indicators to manage ecosystems (Jamieson, 1992). The concepts of soil quality and ecosystem health are somewhat vague or even arbitrary, but can serve as a means to combine scientific consensus with social values to effectively manage soils and ecosystems.

B. Health as a measure of fitness

Analogies between human health, as studied by physiology, and ecosystem health suggest that indices of ecosystem condition or stress may be used as a guide to comprehend and manage ecosystems. Such indices might provide an assessment of ecosystem function in the same way body temperature and blood pressure measure the physiological status of humans. Alternatively, the indices might be viewed as set points or goals sought by the human controllers of managed systems. Since health or physiological assessments focus on functions and vital processes of living systems, a transdisciplinary or ecological approach founded on physics, chemistry, and biology is essential. As an inherently transdisciplinary field, soil science is well equipped to help evaluate ecosystem health, provided that efforts are oriented more toward ecosystem function (e.g., material and energy transformations) and structure (e.g., information linkages, architecture, biodiversity), rather than crop production alone. To assess the health of soil sub-systems and linkages between soil and ecosystem health, Rapport et al. (1997) suggested that groups of several indicators are more useful than single indicators, and that different categories of indicators may be required. Some categories consist of indicators that are most useful to diagnose the cause of dysfunction, while others are most useful to assess overall health or potential risk.

Various definitions for ecosystem health have been proposed. In the introductory chapter of *Ecosystem Health*, the editors propose an adaptive, experimental approach to environmental management, an approach that incorporates both ecosystem function and social values (Haskell et al., 1992). They suggested that a healthy ecosystem is free of distress and must be stable and sustainable; that is, the ecosystem must actively maintain its organization and autonomy over time and retain its resilience to stress. The NRC (1994) in the U.S.A. also emphasized sustainability when rangeland health was defined as "*the degree to which the integrity of soil and the ecological processes of rangeland ecosystems are sustained.*" The report distinguished between healthy and at-risk rangeland (separated by an early warning line), and between at-risk and unhealthy rangeland (separated by a threshold of rangeland health).

Concepts of soil or ecosystem quality and health (and perhaps even sustainability, integrity, resilience, and fragility) may be lumped together on the grounds that both encompass fitness, function, and sustainability. Such concepts will have practical value if they help guide ecosystem management to maintain structure and function while satisfying social goals. Assessments of quality or health require some standard of comparison or benchmark against which system function is assessed. What exactly constitutes a good quality soil or a healthy ecosystem is debatable, but must be stated clearly at the outset of any assessment.

C. Natural versus agricultural systems

The benchmark most widely used to assess the relative health or quality of agricultural soil is soil under natural or unmanaged ecosystems, often in the same locality, with similar parent materials, topography, and climate. Agroecosystems differ from natural ones in several respects (Table 5.1). Advocates of "ecological agriculture" suggest that natural systems in the locality are the most appropriate structural model for agroecosystems (Soule and Piper, 1992). Natural ecosystems sometimes are assumed to attain a state of quasi-equilibrium in which production equals respiration, whereas agroecosystems are maintained in a state of perpetual secondary succession in which production exceeds respiration, with the excess production available for harvest and export. Thus, the functioning of agroecosystems may be fundamentally different from "natural" ones. The challenge then might be either to partition the landscape between contrasting ecosystem types with specific goals (e.g., ranging from aesthetic retreat to food production), or to manage each ecosystem to satisfy multiple goals (i.e., provide an array of services).

Commercial or industrial agriculture tends to replace the stabilizing interactions of natural systems with high-energy inputs, culminating in undesirable surprises, such as pollution, pest resistance, and soil erosion (Edens and Haynes, 1982). In the global N cycle, the amount of human-driven N fixation (mainly industrial N fixation during manufacture of fertilizers) now exceeds that fixed by natural processes (Vitousek,

TABLE 5.1

General characteristics of agricultural and 'natural' ecosystems

Characteristic	Natural ecosystem	Agroecosystem
Human impacts	Indirect; although sometimes managed (e.g., parks)	Direct; intentionally managed (human 'controllers')
Energy source	Sunlight	Sunlight and fossil fuels (i.e., subsidized by previous sunlight)
Internal nutrient recycling	Higher; plants depend on nutrient mobilization in soil	Lower; plants become dependent on exogenous nutrient inputs
Nutrient balance	Inputs ≥ outputs; nutrients retained from soil and atmospheric inputs	Inputs ≤ outputs; nutrients exported with harvested biomass
Spatial complexity	Patchy landscape; multiple niches patchy organism distribution	Vast homogeneous tracts of land uniform organism distribution
Temporal complexity	Sporadic ecosystem dynamics; processes recur at low frequency	Uniformly managed dynamics processes recur at high frequency
Biodiversity	Higher; influenced by ecological interactions and humans	Lower; dictated by crop selection and agronomic management
Plant genetics	Native perennials that have adapted or evolved *in situ*	Annual cultivars, often hybrids, selected and manipulated by humans
Soil disturbance	Infrequent; caused by severe storms, burrowing soil fauna	Frequent; caused by tillage, land levelling, land drainage
Chemical signals	Hormones produced by plants, soil organisms and animals	Pesticides applied to 'control' pests, synthetic growth hormones
Mutualistic interactions	Probable; occur when coevolution conveys a competitive advantage	Improbable; often are unconsciously eliminated to simplify management

1994). Fossil fuel combustion and N fertilization have increased both atmospheric CO_2 concentrations and deposition of fixed N. This eutrophication of the ecosphere, a consequence of energy consumption by humans, is expected to eliminate species and impair ecosystem function (Schlesinger, 1994; Ehrlich, 1994).

The far-reaching influence (in both time and space) of humans on ecosystems may hamper assessments of soil quality and ecosystem health, because natural systems are unavailable as benchmarks. Fires set by prehistoric humans continue to shape present day ecosystems (Westbroek et al., 1993). Management (often inadvertent) of so-called natural areas, including fire suppression and human encroachment in parks, also has eliminated ecosystems that have minimal human influence (Wagner and Kay, 1993). These observations suggest that humans are, in fact, a natural component of ecosystems, and that successful management strategies might at least consider, but not be restricted to, human values and interactions among humans and other ecosystem components.

D. Broad-scale indices

Various indices of soil quality have been proposed (refer to other chapters in this volume), but few assess functioning at the ecosystem scale. Rather than assessing specific soil functions, an ecosystem perspective requires evaluation of general or fundamental functions.

Schlesinger (1994) proposed a set of indices that encompass interactions among biota and soils, and have general applicability (Table 5.2). Net primary productivity less external inputs of energy measures the capacity of the system to capture solar energy. In addition to solar radiation, external inputs of energy in the form of fertilizers, irrigation, tillage, and pesticides usually are required to maintain the flow of food and fiber from agroecosystems and to maintain the masses of soil organic matter and biogeochemicals. Stable levels of soil organic matter imply that the most dynamic portion of the soil is being maintained. Biogeochemical mass balance and internal nutrient cycling indicate how well the ecosystem retains nutrients and how efficiently these nutrients are utilized. Transformations in the soil are major determinants of nutrient retention and internal recycling. Similar criteria were proposed to assess rangeland health (NRC, 1994), but in addition, the presence of properly functioning recovery mechanisms was deemed essential to ecosystem health (Table 5.2). Recovery mechanisms or stress resilience generally involve biotic modification of the abiotic environment in response to extreme events such as fire and drought.

TABLE 5.2

Broad-scale indices of soil quality and ecosystem health

Ecosystem index	Desirable, healthy or high quality condition
Adapted from Schlesinger (1994):	
1. net primary production – external energy inputs	high, given climatic constraints
2. soil organic matter	static quantity
3. biogeochemical mass balance	inputs $\geq$ losses
4. internal ecosystem nutrient recycling	recycling $\gg$ external inputs
Adapted from NRC (1994):	
1. soil stability and watershed function	resistant to erosion and runoff
2. nutrient cycling and energy flow	high nutrient retention and energy capture
3. recovery mechanisms	unimpaired recovery mechanisms
Adapted from Schneider and Kay (1994):	
1. energy dissipation	high
1.1 energy capture and internal flow	high, greater potential for dissipation
1.2 element cycling	more cycles, longer cycles, slower rates
1.3 trophic structure	longer, more efficient food chains
1.4 respiration and transpiration	high
1.5 ecosystem biomass	large
1.6 biodiversity	high, diverse pathways for dissipation

The complexity and organization of living systems, which seem to defy the second law of thermodynamics (i.e., increasing disorder and maximization of entropy), may also provide a means to assess ecosystem function at a broad scale. This organization has been ascribed to the ability of nonlinear open systems, operating far from equilibrium (i.e., with external energy inputs), to dissipate energy (Prigogine et al., 1972). Organization emerges spontaneously in living systems in response to inputs of useful energy. Schneider and Kay (1994) suggested that these energy inputs tend to displace systems from steady states, but organization generated within systems dissipates the energy. As energy inputs increase, more organization emerges to dissipate the energy, and displacements from steady states opposed. Thus, ecosystem health may be assessed as the capacity to dissipate energy: ecosystems that are more developed, unstressed, and healthy will have a greater ability to degrade or dissipate energy.

The various measures of energy dissipation suggested by Schneider and Kay (1994) are consistent with other indices which were not formally based on thermodynamic principles (Table 5.2). Terrestrial ecosystems, especially vegetated ones, dissipate the majority of incoming solar energy by evapotranspiration and respiration. Thus, given the same energy input, the healthiest or at least the most developed ecosystem would be expected to re-radiate the least amount of useful energy (i.e., to have the lowest black body temperature, possibly measured by remote sensing).

In accordance with Schneider and Kay's (1994) thermodynamic interpretation, Lotka's (1922) maximum power principle (MPP) suggests that self-organization within ecosystems will maximize the flow of useful energy and increase the organic mass of the system. Thus, soil organic matter accumulation may be viewed as the balance between storage of useful energy as plant residues in soils and dissipation of this energy when soil organisms decompose the residues to catalyze the inflow of additional residues and energy (Anderson, 1995). To increase the total energy flux through the system, as much soil organic matter and useful energy are maintained as possible, but runaway accumulation nearly always is prevented by other constraints, such as nutrient or moisture availability. In an analogy used by Lotka (1922) the MPP suggests that agroecosystems may be viewed as a faster-turning wheel, rather than a larger wheel, compared to those of natural systems.

Non-equilibrium thermodynamics also formed the basis for Addiscott's (1995) discussions of ecosystem sustainability, only his interpretations were based on the premise that minimum production of entropy is a criterion of sustainable systems. Sustainability depended on a proper balance between entropy-producing and entropy-lessening processes or between dissipative and ordering processes. Small molecules, such as CO_2 and NO_3, are the result of entropy-producing processes that degrade complex, ordered structures, such as soil organic matter. He proposed that an audit of small molecules might be used to evaluate the sustainability of contrasting agroecosystems.

Several other indices may be useful to gauge ecosystem health and soil quality. In addition to plant growth and levels of soil organic matter, Pankhurst (1994) suggested that the diversity, species composition, mass, and activity of soil organisms might prove useful as indicators of soil productivity and sustainability. Consumers, such as higher-level grazers in detrital food chains and cattle on rangeland, often maximize the energy flows in ecosystems (Loreau, 1995). Excessive consumption by grazers or predators of groceries in the supermarket, however, may break nutrient cycles and decrease energy flow and organic matter. By increasing the proportion of plant production that is exported from agroecosystems (i.e., harvest index), humans have increased consumption, with a concomitant decrease in the return of organic matter to the soil. The direct dependence of human well-being on ecosystem health is conspicuous when health is measured as the ability of ecosystems to provide goods and services to humans (Pulliam and Haddad, 1994).

VII. SUSTAINING HUMANS

Indices of ecosystem health and soil quality draw attention to the flow of food and other services that must be sustained by the ecosystem or soil, without impairing the stability or resilience of the system. The implications of human population for ecosystem health cannot be ignored, because the flow of food and other services obtained from ecosystems is determined by sheer numbers of humans and their rates of resource consumption (exacerbated by over-consumption of resources by affluent humans) (Shaw , 1989; Pulliam and Haddad, 1994). The concept of human carrying capacity requires information on the maximum flow of ecosystem services that may be obtained without impairing soil quality and ecosystem health (Kessler, 1994).

Loomis and Conner (1992) are optimistic that current technology might support 20 billion humans (10 billion on the current area of farmland), but fertilizer and other external inputs will be essential. Others agree that vast increases in global food production may be possible, but they remain pessimistic that increases will come without enormous environmental costs (e.g., loss of species and other non-food services from ecosystems) that will severely degrade the quality of human and likely all life. By allowing us to obtain services more efficiently, technological advances often are endorsed as solutions to ecological degradation (Ausubel, 1996). The dependence of humans on the flow of life-support services from ecosystems, however, might suggest that human ingenuity and technology are inadequate substitutes for natural capital (Rees, 1995). This suggests that technological advances, especially those used in food production, must be carefully integrated to work in accord with ecosystem processes rather than used to substitute for ecosystem processes. Since human carrying capacity is influenced by both biological evolution and human innovation, it is dynamic and very difficult to estimate. Consequently, ecosystem resilience (i.e., ability of the system to withstand disturbances) may be a more useful index of sustainability than proximity to some uncertain carrying capacity (Arrow et al., 1995).

In view of the poorly understood interactions among ecosystem components and the potential for chaotic dynamics, it is unrealistic to expect any set of parameters to definitively measure ecosystem health or soil quality. Since prehistoric times humans have had considerable impacts on their environment (Westbroek et al., 1993; Wagner and Kay, 1993), and our impacts inevitably will continue to shape both natural and agricultural ecosystems. Consequently, ecosystems that are free of human influence are unavailable as ecological benchmarks. Management of agroecosystems is deemed successful when production is maximized and deviations from maximum production are controlled or eliminated. But change, possibly change that cannot be anticipated, seems inevitable. Perhaps a greater appreciation for interactions among ecosystem components will provide the insight necessary for humans to act within ecosystems in cooperation with other life, rather than as external controllers.

VIII. SUMMARY AND CONCLUSIONS

This chapter is intended to further the idea of soil quality from an ecosystem perspective, to foster scientific study and debate, as well as to obtain input on the proper goals for ecosystem management, particularly the management of agroecosystems. Soils are considered as components that influence ecosystem structure and function across a wide range of spatial and temporal scales. The contribution of soils to ecosystem function (especially energy and matter transformations) and structure (architecture, information linkages, food webs, biodiversity) is assessed from the standpoint of systems dynamics. Soils are nested within larger ecosystems and are involved in a myriad of interactions that contribute to ecosystem complexity. Ecosystems are cybernetic, that is, they exchange information among system components and are self-regulating. No discrete regulator or controller is involved in natural systems, but humans are the controllers in agroecosystems, something like a governor might control the speed of an engine. Since humans are inextricably linked to ecosystems, cooperative and reciprocal behaviour which recognizes interactions among humans and other ecosystem components may be more desirable than altruistic behaviour which too often is undermined by deceit (Tullberg and Tullberg, 1996).

Ecosystem development may reach equilibrium as a result of positive and negative feedback among information linkages. The equilibrium, however, is dynamic, because ecosystems may acquire the following properties: resilience about the equilibrium state, multiple stable states separated by thresholds, and chaotic dynamics. Ecosystem interactions, complexity, and dynamics suggest that broad-scale or macroscopic indicators of soil quality and ecosystem health may be most informative. Proposed indicators include energy capture or dissipation and biogeochemical cycling or retention, but selection of an appropriate benchmark for comparison may be problematic. The anthropocentric bias of any assessment must be stated explicitly, because a single index likely will yield an inadequate assessment of the full range of life support services required of soils and the ecosystems within which they are embedded.

Almost four decades ago, Nikiforoff (1959) recognized that soil science and agronomy had been diverted from a promising start on basic research from a pedological perspective to deal in an empirical way with the pressing problems of crop production. In discussing the future prospects for a more productive agriculture to supply an ever-increasing human population, he wrote, "*This means a rather drastic meddling with the short geochemical or pedogenic cycles and might lead to*

TABLE 5.3

Differences between ecosystem and more traditional reductionist perspectives of various factors influencing crop production

Factor	Ecosystem perspective	Reductionist perspective
Soil	Part of a larger ecosystem	Independent resource or input
Soil function	Wide array of life-support services	Support crop production
Cause and effect	Non linear	Linear; directly proportional
Stability	Dynamic, multiple equilibria resilient and adaptable	Static, single equilibrium resistant to change
Predictability	Fundamentally random	Deterministic
Role of humans	Internal cooperators	External controllers
Human values	Partial influence, stated explicitly; motivate conservation	Over-riding influence, implicit exclusively anthropocentric
Management focus	Interactions between humans and ecosystems	Resource exploitation
Management goals	Multiple, including production, conservation, aesthetics	Production oriented
Research orientation	Possible pathways to sustainability; integrative and adaptable science	Preconceived optimum practices; discrete technology

disastrous results if it is undertaken without a thorough understanding of the process." A reappraisal of soil functions and of the role of humans within ecosystems might help to avoid at least some disastrous results. Several differences exist between our ecosystem perspective and more traditional, often reductionist, perspectives that have helped to increase agricultural production in the past (Table 5.3). Vandermeer (1995) has a valid concern that multi-dimensional complexity may render a rigid systems-ecology approach "hopelessly grand" or impractical. To avoid the pitfalls of reductionism, however, now might also be a good time to step back and take a careful look at the big picture.

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Chapter 6

SOIL QUALITY INDICATORS: PEDOLOGICAL ASPECTS

R.J. MacEWAN

I. INTRODUCTION

A. Overview

The notion of "soil quality" demands from its users the development and clarification of a new paradigm, and further, it requires reconsideration of approaches to soil description, quantification, and classification. Pedologists are concerned with understanding the variety of soils, their distribution, and genesis, and should therefore be directly involved in clarifying and explaining soil quality. An approach to redressing the philosophical imbalance between recognition of soil quality (a synthetic or holistic notion) and the selection of single properties to serve as indicators (a reductionist approach) is offered in this chapter. Neither the former nor the latter has exclusive virtues; rather, the broad holistic view must be maintained while specific tools are applied within this branch of environmental assessment.

Although functional properties may be measured uniformly across all soils (e.g., comparison of available water capacities, internal drainage, structural stability), the diversity of soils and their related properties militates against the use of blanket class groups for monitoring *changes* in soil quality in a uniform way across all soils. Inherent pedogenic characteristics that are unique to different soils dictate the way in which any selected indicator should be interpreted in each case. All soils are not equal in relation to their potential for structural development, nor are they equally resilient to environmental influences; indeed, this resilience changes throughout the life of the soil. The pedological context for soil quality demands an approach that accounts for variability of soils, not only in space, but also in relation to time scales ranging from thousands of years down to short cycles of land management.

B. Soil quality as a factor in land settlement

The development of crop husbandry approximately 10,000 years ago launched us towards an inevitable encounter with the paradigm of soil quality. Prior to this there was no need to understand the differences that exist below the ground surface. The dependence on crop surpluses to feed expanding populations, who are mostly not engaged in any primary food production or collection, is one that focuses on the ability of the soil to support plant growth. Selection of plant varieties and choice of land use, type of tillage, mode of irrigation, and frequency and type of crop rotation are all decisions made in relation to differences in soil quality. These differences are recognised primarily through the effort required for tillage, the availability of water, and the plant response. Consequently, in post-neolithic cultures, written records and the development of land use patterns all stand as testimony to the recognition that soil is not a uniform material but varies from place to place. The variety of soils and their associate properties precondition the success of selected land uses and agronomic practices.

Hence, soil suitability and land capability, which are inherent aspects of soil quality, are not new ideas, though they may be regarded as recent sciences. Krupenikov (1993), reviewing the history of soil science, cites Chinese, Egyptian, ancient Greek, Roman, Indian, and Mayan examples that all show awareness of soil differences and their corresponding management requirements. The adaptation of the plow in different forms has also been closely related to differences in soil properties. In ancient Mesopotamia two types of plow were used: a light plow, drawn by asses, for light-textured, shallow soils, and a heavy plow, drawn by bulls, for the heavy-textured, clay soils (ibid., p. 19). Hoskins (1988) has given a comprehensive account of the historical development of land use patterns in England and related these as much to soil differences as to legislative and engineering considerations.

C. Modern agriculture, soil quality, and environmental planning

During the industrial revolution, steam-driven traction engines were applied to agricultural problems and assisted the commissioning of more intractable lands and soils, particularly encouraging drainage of wet land. The advent of the affordable petrol-driven tractor after the 1930s enabled mechanisation of farming practices generally, and divorced the operator from the close contact with the soil that would be experienced in toil behind implements drawn by horse or bullock. Trends in the U.K. towards increasing field size over the last forty years, particularly in the 1960s, have entailed the removal of hedges, which were historical divisions often based on soil type. More powerful mechanization of agriculture has increased efficiencies in tillage, sowing, and harvesting operations, but this has also resulted in uniform practice and timing of operations on larger management units containing greater inherent soil variability than before.

Consequently, advances in agricultural machinery have had two contradictory effects. First, they have created an artificial independence of, or at least an insensitivity to, soil variability. Secondly, they have precipitated an urgency to understand management needs of different soils because of variations in yields across management units and declining soil quality of vulnerable soils.

In Australia, where most cropping land was developed using powerful machinery during the middle of this century, cadastral boundaries were not defined by soil or even land type, although the occupation of land for agriculture has followed some land–vegetation associations. The allocation of surveyed square mile blocks was based on office, flat map sheet perspectives and has resulted in land use patterns that bear little relation to natural drainage conditions or soil variability. Insensitivity to soils and hydrology has had a number of undesired consequences for land holders, such as flooding, waterlogging, and increased salinisation of land. In addition, the trend towards larger, heavier farm equipment has caused compaction problems on most cropped soils. Soil-related land degradation problems in Australia given prominence in the Commonwealth Government's Decade of Landcare plan, are soil acidification, soil structure decline, soil contamination, water repellency, wind erosion, water erosion, waterlogging, dryland salinisation, and irrigation salinity (Commonwealth of Australia, 1991). Identification of these issues, or processes, is in common with other countries, such as Canada (Acton and Gregorich, 1995).

II. PEDOLOGY AND THE SOIL QUALITY PARADIGM

The paradigm of soil quality includes both static ("potential") and dynamic ("function–process–response") frameworks. The former has been well developed in pedology and land evaluation (Dent and Young, 1981), whereas the latter has had sparse attention and is still being developed. Soil maps, soil reports, and soil classification systems are useful tools in planning the use and management of land. However, their utility depends on the original purposes set out in their production, the scale of investigation, the class limits used to define boundaries between mapping units, the parameters included in the classification scheme, and the manner of data storage. Traditionally, they tend be "static" in their representation of reality, allowing for allocation of land use on the basis of capability and suitability ratings, but offering no real opportunity for ongoing monitoring of soil condition or changes in soil quality. However, management of soil and land data within a G.I.S. (Geographical Information System) will permit time-series analysis in soil quality monitoring.

A. Pedology: definition and scope related to soil quality

Pedology has two broad purposes: to describe and classify soil differences, and to interpret them with respect to their management requirements. A definition of pedology provided by Wilding (1994) is "*that component of earth science that quantifies the factors and processes of soil formation including the quality, extent, distribution, spatial variability, and interpretation of soils from microscopic to megascopic scales.*" This definition introduces the word "quality" in a general way, and, as in many other instances, its usage is indefinite. Soil Survey Staff (1951) use the term *soil quality* loosely throughout the soil survey manual, usually to imply some sense of the soil's value.

At the simplest level we can interpret quality as a soil property, as in "red soil" and "stony soil", or we can go further and interpret these properties or qualities in relation to soil use on a ranked scale from good to poor soil quality. Such categorisation of soils can be considered as recognition of the more enduring aspects of soil quality affecting land capability and soil suitability (Dent and Young, 1981). Monitoring of the soil response to management, through measurement of functional properties, can be carried out to ensure that soil quality is not compromised by inappropriate choices in land use or level of management.

If we emphasise soil quality more explicitly in a definition of pedology, we could rewrite Wilding's definition above as "*that component of earth science that quantifies, at all scales, the factors and processes of soil formation, including the extent, distribution, and spatial variability of soils, and provides frameworks for interpretation and classification that can be applied to management and protection of the soil resource and its functional qualities.*"

B. The need for functional classification of soils

Soil classification systems are potentially important management tools but have, in general, been poorly or only obscurely related to soil quality. They are our attempts to bring conceptual order into the complex world of soils and to allow knowledge gained in one location to be used in another. The task is large because soils are changing, their evolutionary history is only partially understood, their future is dimly perceived, and they are used for a range of purposes all with unique requirements in relation to land use and soil function.

The soil science literature, particularly in the 1950s and 1960s, has many examples of rationales for classification. An excellent selection of papers on soil classification was compiled by Finkl (1982), and the debate continues. In Australia at this time three systems are in common usage: Great Soil Groups (Stace et al., 1968), the Factual Key (Northcote, 1979), and Soil Taxonomy (Soil Survey Staff, 1988), and yet a further system has just been published (Isbell, 1996).

Soil classification is traditionally based on three different, but complementary, approaches: a *zonal* approach, tackling classification on the basis of climate as the overriding soil-forming factor and answering the question "Where was the soil formed?"; a *genetic* approach, posing the question "How was the soil formed?"; and a *morphological* approach, adopting the pragmatic view that we are always guessing pedogenesis and should group soils after answering the question "What does the soil look like?" Paton et al. (1995) have revived the debate concerning these three approaches in pedology and present a sound case for a more thorough application of studies in pedogenesis.

So far we have not rigorously applied soil classification to the scrutiny of investigation involved in posing the question "How does this soil function?" Approaching soil classification in this way would place zonalism in context (temperature and rainfall influences on current functions), use morphological features to interpret current processes, and apply theories of pedogenesis to account for current soil quality or to predict likely future developments within the soil and landscape.

Functional soil qualities as defined by Carter (1996) and Carter et al. (this volume) could be used as a basis for classification and would provide more utilitarian distinctions between soils. Indeed, Bouma (1994) has questioned whether different soil taxa are also different in terms of function and suggests that, if they are not, then groupings of different soil taxa might be appropriate for the development of pedotransfer functions. Wösten et al. (1985) showed that the number of soil units could be reduced by a factor of three when water supply capacity was made the focus of a functional analysis. Partitioning of water and solute flow was investigated by Seyfried et al. (1992) in relation to soil taxonomic classification of 689 Florida pedons. They found that USDA Soil Taxonomy was effective at the order, suborder, and great group levels in discriminating between quality of soils for this function and concluded that lumping of lower-order categories would be justified.

There is an increasing interest in management of disturbed materials and derelict land (Bullock and Gregory, 1991), and much progress is being made in development of classification systems based on functional quality. Starting with "non soil" and thinking in terms of rehabilitation and soil development naturally encourages a qualitative assessment of materials in relation to their potential as soil parent materials. Hollis (1991) has reviewed this subject and presents a useful table of diagnostic properties for which he proposes functionally based taxonomic divisions such as critical depths for root growth, toxicity threshold values, and erodibility.

C. Soil formation and soil quality

Judgements about soil quality are usually made with the tacit assumption that the soil as encountered is a relatively permanent feature; we don't see what it has been, nor what it is about to become. However, soil attributes and soil quality naturally change with time. In the discussion that follows, the dynamic aspects of pedogenesis in relation to soil quality are briefly reviewed.

1. Factors and processes of soil formation

Factors of soil formation were proposed in the 1860s in the U.S.A. by E.W. Hilgard (Jenny, 1961a) and in the 1880s in Russia by V.V. Dokuchaev (Krupenikov,

1993) and have been developed in a semi-quantitative fashion by Jenny (1941, 1961b, 1980) as the now well accepted relationship:

$Soil = f\{Climate, organisms, relief, parent\ material, time \ldots\}$

Humankind has been considered either with organisms within the 'o' factor or as an additional factor of soil formation (Bidwell and Hole, 1965; Jenny, 1980). Simonson's (1959) general framework for soil genesis, as two overlapping steps of parent-material accumulation and differentiation into horizons, complements and augments Jenny's model. Four groups of processes (additions; removals; transfers, or translocations; and transformations) were described as relevant to the development of horizons. Simonson suggested that changes due to these types of processes take place simultaneously in all soils, but that the different balance of these would result in distinct soil types, such as chernozems and latosols. This theory is widely accepted (Birkeland, 1974; Fanning and Fanning, 1989; Wilding et al., 1983), and Arnold (1983) provides a table of terms used to describe soil formation in which he has provided an indication of which of the four processes dominates in each of the genetic processes named.

However, pedological research still emphasizes the development of what might be considered the more extreme soils, that is, those with characteristics that could be considered end points in pedogenic development, such as podzols. A suite of processes, *erdefication* (from German *erde*, earth), has also been proposed to designate development of A1 and Bw (cambic) horizons with no distinct segregations or accumulations (Conacher and Dalrymple, 1977, p. 67). This addition to the already accepted processes provides the reference paradigm for soil formation per se rather than focusing on diagnostic horizons that are concentrations or losses of major soil constituents.

It is worth considering that where erdefication dominates over other processes, such as podzolization, alkalization, calcification, gleying, etc., soil quality would be higher than in soils where the latter processes dominate. For sustainability of soil quality we need to devise ways of encouraging erdefication or "earth-forming processes" with respect to maintenance of soil structure, humification, and mineralization. Such a task is stated as an ideal, as the relative dominance of additions, removals, translocation, or transformation of soil components is also a function of the soil's relative position in the landscape (ibid., p. 137ff).

2. Dynamism of pedogenesis: the ecological theatre and the evolutionary play

A dynamic approach to pedogenesis, building on the perspectives offered by Jenny (1941, 1961b, 1980) and Simonson (1959, 1968), can be used to provide a framework for the assessment of soil quality at different spatial and temporal scales. An analogy appropriate to our conceptual framework for pedogenesis has been used in ecology by Hutchinson (1965) as the ecological theatre and the evolutionary play. We can envisage the soil-forming factors as setting the stage or conditions for soil formation: the ecological theatre. The play that is enacted within the theatre is the suite of processes interacting over time and modifying the original solum. When we open a soil pit we peek briefly into a moment taken from the play and view the imprint of all

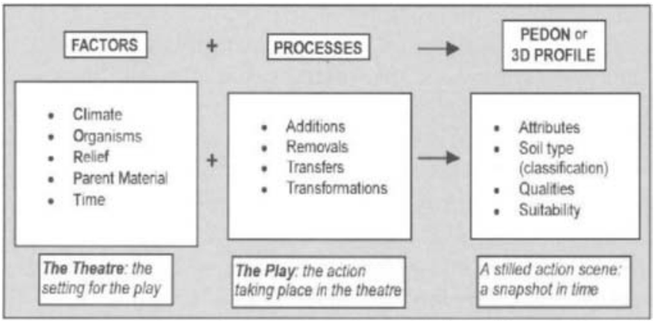

Fig. 6.1. Pedogenesis as the relationship between soil-forming factors and processes. The analogy of the ecological theatre and the evolutionary play: the land manager appears as actor and co-director in metapedogenesis affecting both factors and processes through interventions in soil and crop management.

that has occurred up to that moment. Human intervention in the play is both as actor and co-director, re-defining the plot and changing the scope of the theatre.

Factors and processes combine to affect the overall soil body recognised and described as the pedon (Fig. 6.1). Resulting soil attributes such as colour, texture, structure, thickness, and chemical properties of horizons are used to classify the soil. The same attributes may be grouped selectively to define soil qualities such as permeability and rooting environment. These qualities, in combination with other land attributes and economic considerations, may then be used to interpret soil suitability for particular purposes (e.g., FAO, 1976; 1983; 1985). Monitoring soil changes through selected attributes must be carried out periodically to determine soil quality and provide feedback into management practices, so that the role of "director" in the evolutionary play is carried out consciously and responsibly.

3. Pedogenesis and metapedogenesis

In natural conditions the pace of pedogenesis is barely perceptible within human generation spans. In managed conditions, the pace and direction of pedogenesis can be altered through engineered effects on the soil-forming factors and processes. These human induced changes in pedogenesis have been referred to as *metapedogenesis* (Yaalon and Yaron, 1966). In assessing the impact of land use practices the metapedogenic influences need to be distinguished from the 'natural' rate of pedogenesis. The former may be beneficial, improving soil quality (e.g. drainage), or harmful, resulting in soil degradation (Bidwell and Hole, 1965; Fournier, 1989). The sustainability pathway, minimising entropy and maximising system productivity, avoids or postpones *pedonemesis*.

Thermodynamic principles can be applied to assessment of soil quality in relation to pedogenic time scales through consideration of the total energy and entropy of the soil–landscape–management system, but examples of such an approach are limited. Smeck et al. (1983) addressed the question of entropy and soil dynamics, and Ibanez et al. (1995) considered ecosystem richness in measures of pedodiversity. Both groups of authors focused on the soil horizon as the element representing order or complexity in the soil profile. They contrasted disordered or simple soil, having no horizonation, with ordered or complex soils, having well developed horizons. Such approaches are limited by scale and the lack of any energetic criteria—why should soil horizons be equated with thermodynamic complexity? They exist at a scale that, apart from A1 horizons, represents dissolution, sorting, and energetic simplification of the soil parent material. A soil with a greater number of soil horizons is not necessarily a good indicator of low entropy; rather, the converse is true. There is, therefore, considerable scope for improvement of thermodynamic frameworks for assessment of sustainability of soil quality.

Metapedogenic influences involve inputs of large amounts of energy and small amounts of matter, decreasing the entropy of some soil components (e.g., addition of nutrients) but increasing the entropy of others (e.g., increasing leaching, accelerating erosion). Soil quality indicators could be selected to monitor soil functions and to measure the entropic status and energetic demands of the system for maintaining those functions. Such an approach would allow a genuine assessment of sustainability of the management systems with respect to different soils and the inputs needed to maintain soil quality.

The capacity of the soil to recover from disturbance in short time frames is related to its inherent and dynamic properties, and so this capacity too must change over time. The term *soil resilience* has been used by some authors to refer to this capacity in a general way (Szabolcs, 1994) or in relation to an attribute such as structural aggregation (Kay, 1990), while others have qualified soil resiliency in terms of soil functions (Blum and Santelises, 1994) and the degree to which the soil will return to functional capacity after disturbance. Soil resilience and soil quality are both highly dependent on pedogenic age and are not necessarily equivalent. Soil resilience is the aspect of soil quality that reflects the ability of a soil to recover its qualitative functions after disturbance. The level of recovery of soil quality depends on soil resilience and soil management, with the latter being more significant in less resilient soils.

D. Spatial scale, pedogenic processes, and soil quality

The need to assess soil quality at different temporal and spatial scales has been widely discussed (Bouma and Bregt, 1989; Carter, 1996; Larson and Pierce, 1991). The question of spatial scale in pedological research is effectively summarised by Dijkerman (1974), who suggests an organisational hierarchy, based on size, of seven subsystems for soil studies. This approach is supported by Sposito and Reginato (1992) and developed by Hoosbeek and Bryant (1992), who describe the role of different types of models at a range of scales.

The pedon (Soil Survey Staff, 1988) is accepted as the basic three-dimensional unit of soil encompassing the variations in horizon and profile features that would fully characterise the soil type under investigation. The pedon exists in the larger subsystem hierarchy of polypedon, toposequence, and catchment or region, and contains smaller subsystems of horizons, peds, mineral organic complexes, and minerals. Investigation of soil at the pedon scale should always include details at the horizon scale and context at the catena or catchment scale (soil–landscape); it is naturally observed as a sample representing the larger polypedon, paddock, or management unit and is not seen as an isolated entity.

Techniques for description and quantification of attributes of subsystems, including their spatial variability, are reasonably advanced at all scales (Bouma and Bregt, 1989; Burrough et al., 1994; Wilding and Drees, 1983). Interpretation of features within subsystems in relation to their significance to management and production systems is also possible, particularly at the pedon (Northcote, 1983) and polypedon/catena (Pennock et al., 1994) scales. Soil quality assessment can therefore be related to any scale of pedological interest.

Temporal variability of soils is less well documented than spatial variability (Burrough et al., 1994). Although there is a body of knowledge that can relate gross changes that have occurred in soil morphology and processes between soils in their native state and soils under intensive agricultural management (e.g., Quideau and Bockheim, 1996), the changes are most extreme within the first few years after establishing agriculture (e.g., Pennock et al., 1994). Most research into rates of pedogenesis is based on time scales that confound geomorphic changes with soil development (Vreeken, 1975), and the indices developed are coarse (e.g., Dorronsoro, 1994; Langley Turnbaugh and Evans, 1994). Schaetzl et al. (1994) advocate functions that are more elaborate than simple linear or logarithmic data-fitting methods for chronofunction modelling. Our current understanding of pedogenic rates is therefore largely generalised over time periods that extend significantly beyond the time frames relevant to managers of soil resources, although there are some cases where pedogenesis is known to be extremely rapid (e.g., Prosser and Roseby, 1995). Arnold et al. (1990; cited by Larson and Pierce, 1991) have tabulated the time changeability of various soil characteristics for the pedon/horizon/mineral scales. They divided changes into three categories and emphasised the need to distinguish between trend changes, relevant to monitoring, and cyclic and random changes, affecting timing of monitoring and noisiness of data.

Hoosbeek and Bryant (1992) reviewed progress towards the quantitative modelling of soil processes. They characterized models with respect to their relative degree of computation (qualitative to quantitative), complexity (functional to mechanistic), and level of organisation (microscopic to megascopic). They concluded that although qualitative models have aided in soil survey and understanding of soils in landscapes, there is a need to devise quantitative models to predict how soils will change in future. This is particularly important if soil quality and land quality indicators are to be used to judge sustainability, and not merely suitability, of current management practices. Bouma and Hoosbeek (1995) and Bouma (1996) have demonstrated a scheme for the integration of spatial scale, model complexity, and level of knowledge applied (Fig. 6.2). It remains for soil scientists working in different disciplines, and at different scales, to develop the links that allow more general application of such approaches.

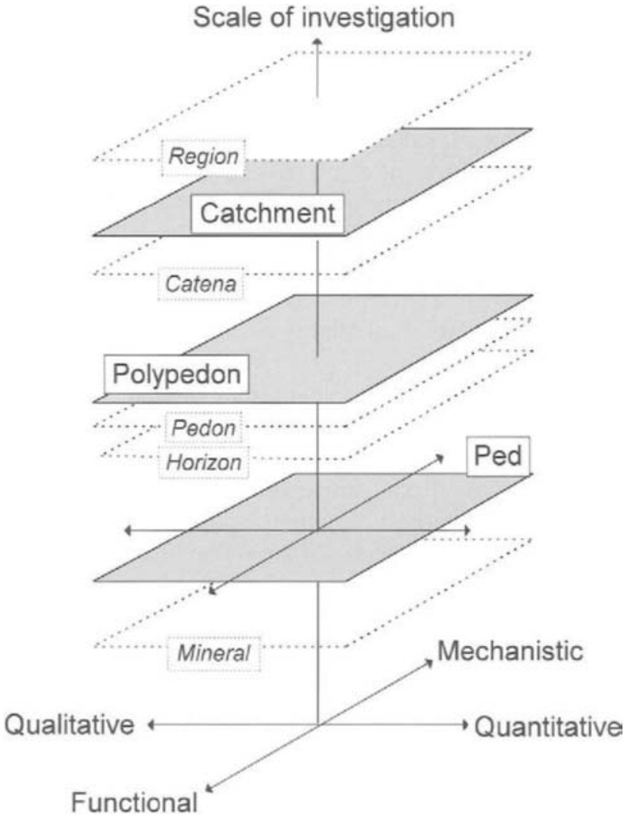

Fig. 6.2. Diagrammatic representation of hierarchy of scales of investigation and model complexity (after Bouma, 1996; Hoosbeek and Bryant 1992;). Quadrants represent models in terms of degree of complexity (functional to mechanistic) and computation (qualitative to quantitative). Highlighting of three levels shows a hypothetical example for integration of soil structure and erosion research at ped, polypedon, and catchment scales.

III. PEDOLOGICAL INDICATORS, PEDOGENIC PROCESSES, AND SOIL QUALITY

A. Criteria for selection of indicators

Appropriate indicators for assessment of soil quality have already been suggested by other authors (Doran et al., 1994; Greenland and Szabolcs, 1994; Larson and Pierce, 1991). A rationale for indicator selection is provided by Casley and Kumar (1987), who propose that indicators should be: (1) unambiguously defined, (2) consistently applied, (3) specific to identifiable qualities, (4) sensitive to change, and (5) easy to collect or measure. How well do common pedological properties satisfy these criteria?

1. Unambiguous definition

This requires an agreed set of standards. In pedology, standards for soil description are subjectively applied with the guidance of nationally accepted handbooks (Hodgson, 1976; McDonald et al., 1990; Milne et al., 1995; Soil Survey Staff, 1993). Although mostly qualitative, there is reasonable international agreement on which parameters should be assessed and on methodology.

2. Consistent application

Soils may be assessed at different seasons, moisture contents, and management stages, so comparison of soil condition over time requires interpretation. Not all parameters have equal relevance to all soils. For example, the assessment of structure of a highly reactive clay requires a different framework to that required to assess structure in a sandy loam or loamy sand, and the two are not comparable. Different soil textures also have different potential for crust formation or development of hard setting horizons, especially when confounded by sodic properties in saline or non-saline conditions (Sumner, 1995). At present there is insufficient development of classes that would be appropriate for sensitive monitoring within different soil types, all soils being currently assessed in the same way.

3. Specific to identifiable qualities

Basic soil morphological data on colour, texture, structure, and rooting depth and frequency can be readily interpreted in relation to the physical quality of the soil for providing conditions for plant growth, with respect to available water capacity, drainage status, and impeding layers (Fitzpatrick, 1996; Northcote, 1983; Wösten, this volume).

4. Sensitivity to change

Many soil properties that are relatively insensitive to management determine the inherent quality of the soil (e.g., profile depth, drainage status, texture, stoniness). Other properties are sensitive to management but also sensitive to season, vegetation cover, wetting and drying, and freeze/thaw cycles. There is also a confounding difficulty in relating major pedological attributes to recent changes due to the lag time between environmental process and system response. Some indicators may be sensitive to change in one direction only; for example, soils that have been drained may still display gley features (James and Fenton, 1993).

5. Ease of measurement or collection

Collection of soil morphological data is time- and labour-intensive but requires only a low level of technical support. Because techniques can be taught to farmers and other land managers, pedological attributes are potentially very relevant for use as indicators.

B. Selection of indicators

Pedogenic processes may be controllable or uncontrollable, and reversible or irreversible. In monitoring soil condition these distinctions are important, and indicators should be selected that reflect controllable processes.

Relationships between scale of investigation, pedogenic processes recognisable at each scale, an indication of whether the processes can be considered as controllable or reversible (by a land manager), their sensitivity to change in years, and the indicators of these processes, are suggested in Table 6.1.

As scale is decreased to cover increasingly large areas, the choice of relevant pedological indicators is reduced, soil data become generalised and sustainability indicators are more relevant to land quality than soil quality. Mapping units cannot represent a single kind of soil at scales smaller than 1:50,000, that is, regional and higher (Dent and Young, 1981) so consideration of soil quality has to be represented by class ranges for single attributes such as pH. Data collected at larger, more detailed scales can always be used to develop maps of areas affected by particular quality defects, such as waterlogging, salinity, or acidification at smaller, regional scales. These latter are the effects of landscape position, land use, and rainfall rather than soil type and so allow interpolation and extrapolation to broad areas from specific instances.

Reporting of sustainability indicators at small scales is used for political and economic purposes, providing information for broad considerations in land protection strategies, such as decisions about allocation of funds for research or for on-ground works. The reliability of such assessments depends on the density and quality of data collected at larger scales. These data may be opinions expressed by experienced field agronomists or conservation staff, questionnaire results from land holder surveys, extrapolation of detailed field research to larger areas, or mapped attributes collected by extensive field survey.

Most pedological attributes are recorded in a qualitative (nominal), or semi-quantitative (ordinal), way, with few interval or ratio data (Bregt et al., 1992). Such data are easily obtainable but are useful as indicators only when gross differences exist between the condition of a parameter prior to and following a period of management. Inherent high variability of these properties, even within a relatively pure soil mapping unit (Wilding and Drees, 1983), means that there is unlikely to be any improvement in assessment of soil quality by finding more precise quantitative methods for most field observed pedological attributes. MacEwan and Fitzpatrick (1996), in consideration of soil morphological attributes, have identified the nature of the data (nominal, ordinal, interval, or ratio) commonly recorded at the pedon scale, and sorted these according to their sensitivity to change within the life time of a land manager.

C. *Most favoured indicators*

Table 6.2 sets out suggestions for the most appropriate indicators of soil quality that could be adopted at the paddock/management unit level by the trained land manager. Implications for the use of some of these are discussed below. All of the suggested indicators would also have appropriate management responses to modify soil quality.

TABLE 6.1

Soil-modifying and soil-forming processes: their relationship to spatial and temporal scale, and suggested indicators for their recognition

Scale	Processes	†	‡	Sensitivity (years)	Indicators
Global and	Plate tectonics	U	I	$10^3 - 10^6$	Vulcanism, continental changes
continental	Soil formation	U	I	$10^2 - 10^4$	Degree of soil development
Extremely	Erosion	U	I	$10^2 - 10^6$	Valley form, river development
small scale	Salinisation	U	R	$10^2 - 10^4$	Halophytes
<1:5000000	Urbanisation	C	I	$10 - 10^2$	Loss of agricultural land
Regional,	Soil formation	U	I	$10^2 - 10^4$	Soil types
catchment	Erosion	C	I	$10^{-1} - 10^3$	Gullies, tunnels etc
or catena	Salinisation	C	R	$10 - 10^3$	Area of discharge/salt affected land
Small scale	Acidification	U	I	$10 - 10^3$	Restricted crop and pasture species
<1:100000	Waterlogging	U	R	Seasonal	Area with slow surface drainage
Paddock or	Erosion, deposition	C	I	$10^{-2} - 1$	Surface features
polypedon	Salinisation	C	R	$10 - 10^2$	Discharge features
Large scale	Acidification	C	R	$10 - 10^3$	pH
>1:25000	Waterlogging	C	R	Seasonal	Ponding, pugging, sealing
Pedon	Erosion, deposition	C	R	$10^{-2} - 1$	Pedestals, rills, layering
(3D profile)	Profile development	C	R	$10 - 10^4$	Depth, horizons
Human scale	Salinisation	C	R	$10 - 10^2$	Vegetation response
1:1	Acidification	C	R	$10^2 - 10^4$	pH
	Waterlogging	C	R	Seasonal	Surface features (pugging, seals), colour
	Sodification	C	R	$10 - 10^4$	Soil dispersion in rain water
	Root penetration and water use	C	R	$1 - 10^2$	Depth and pattern of roots vs textures

TABLE 6.1 (continued)

Scale	Processes	†	‡	Sensitivity (years)	Indicators
Horizon (pedon in detail)	Erosion, deposition	C	R	10^{-3}– 10^{2}	Surface features
	OM accumulation/depletion	C	R	1 – 10^{2}	L,F,H.Consistency (hard setting), OC
	Thickening, thinning	C	R	10 – 10^{2}	Native site comparison
	Leaching, acidification	C	R	10 – 10^{2}	pH
	Clay translocation	C	I	10 – 10^{2}	Coatings, turbidity of run off
	Soluble salt accumulation	C	R	10 – 10^{2}	EC, visible crystals
	Carbonate, gypsum accumulation	U	I	10 – 10^{3}	Nodules etc
	Gleying	C	I	1 – 10^{2}	Colour, mottling
	Iron enrichment	U	I	10^{2} – 10^{4}	Iron pans, buckshot, pore linings, Bs
	Compaction	C	R	Seasonal	Ped shape, pores, bulk density, roots
	Loosening	C	R	Seasonal	Ease of tillage, cloddiness
	Root penetration, water use	C	R	Seasonal	Roots vs. pores vs texture
	Animal activity, burrowing etc	C	R	Seasonal	Number/area(vol)
Ped	Aggregation	C	R	1 – 10^{2}	Water stability
	Cementation	U	I	10 – 10^{3}	Consistency, grain coatings
	Slaking§	C§	R§	10^{-4} – 10^{-2}	Crusts, seals
	Dispersion§	C§	I§	10^{-4} – 10^{-3}	Cutans, turbidity
	Compaction	C	R	Seasonal	Pores, ped/clod density
Mineral	Hydration, hydrolysis, solution	U	I	10^{-4} – 10^{4}	%unweathered minerals
	Salts (formation/transformation)	U	I	10^{-3} – 10^{2}	EC, visible crystals (halite)
	Clay formation	U	I	millenia	%clay and 2:1 vs 1:1 layer silicates
	Clay degradation	U/C	I	10^{2} – 10^{4}	Low pH, water logging, bleached colour
	Fe/Mn oxide formation	U	I	10^{-3} – 10^{4}	Colours: red/ yellow (formation)
	Fe/Mn oxide transformation	C/U	R	10^{-3} – 10^{4}	Bleached/grey colour transformation)

where: † Controllable = C; Uncontrollable = U; ‡ Irreversible = I; Reversible = R

§ Slaking is a reversible soil process because aggregation of slaked soil components can be encouraged with organic matter additions. Dispersion, although it can be prevented by flocculating agents, cannot be reversed due to the total disintegration of peds, destruction of soil fabric, and loss of clay. Both slaking and dispersion are controllable processes.

TABLE 6.2

Indicators, observations, and their significance: the most useable pedological indicators of soil quality at the pedon and polypedon, or management unit scales

Indicators	Observations	Processes or functions affected
Surface features	Thin surface crusts, surface seals, smears with cracks	Soil structure decline (topsoil)
	Stoniness, pedestals, rills	Soil loss (incipient topsoil losses)
	Fence line deposition, material washed onto roads	Soil and nutrient loss
	Prolonged ponding of water after rain or irrigation	Soil structure decline (topsoil or subsoil)
	Pugging damage, vehicle wheel sinkage	Waterlogging, soil structure decline
	Bare soil (no plant cover or litter)	Organic matter decline in topsoil. Runoff
	Bare wet soil, death of preferred plant species, oily films in surface water discharge, salt efflorescence when dry	Mobilisation and accumulation of salt, development of dryland salinity
Roots and pores	Frequency and depth of roots	How well the soil space is being used
	Root occupation of visible pores (biopores and fissures)	Water use, nutrient cycling, recharge
	Frequency of visible pores (by horizons)	Drainage and aeration
Horizon boundary	Connectivity of pores between horizons, pans	Root growth, drainage, aeration (whole pedon)
Soil consistence	Air dry strength of soil—allocation to ordinal class in the range rigid to loose. (A surrogate for texture but also affected by other factors, e.g., organic matter, sodicity)	Root growth, water movement, tilth development
Aggregate stability	Aggregate behaviour (each horizon) in pure water: swelling, slaking, dispersion (Emerson test)	Soil structure decline: development of seals and crusts. Infiltration, aeration, drainage Shoot emergence
EC, pH	1:5 soil: water suspension (pH also in $CaCl_2$)	Salinisation, acidification
Colour	High value/low chroma (Munsell), mottling, rusty linings in root channels	Waterlogging, drainage and aeration Iron mobilisation, accumulation and removal
Bulk density	Direct measurement, core method or volume replacement. Calculate relative compaction, total pore space, air-filled porosity at field capacity	Soil aeration and drainage, root penetration

1. Soil surface features

Ground surface conditions can be regularly assessed for very little effort as no excavation or sophisticated testing is required. Soil surface condition can indicate degrading processes such as erosion, compaction and damage due to wheels or hooves, salinisation, lowered infiltration (seals and crusts), and barriers to seedling emergence (crusts). Monitoring of soil surface condition, particularly in relation to crust development or surface seals can readily be carried out by landholders (e.g., Bresson, 1995). Tongway (1994) has described a similar, albeit simpler, approach for monitoring soils in naturally vegetated Australian rangelands where grazing pressure is causing changes in soil surface cover and condition. Mellis et al. (1996) have reported that qualitative observations on the soil surface of a tilled sandy clay loam provided reliable assessments of early stages of surface degradation that were confirmed by bulk density measurements and tension infiltrometry. Surface condition should be assessed with an eye for variability and catchment context. Bresson (1995), reviewing literature on soil crusting, pointed out that assumptions of crust uniformity are too often made in erosion studies and recommends integration of quantitative studies on soil crust formation and soil erosion at field scales that could take account of crust variability.

Soil surface conditions are particularly sensitive to seasonal and management influences. For example, rills and other symptoms of shallow surface soil loss are readily removed by cultivation. In winter, cultivated soil, exposed to freezing conditions, has cloddiness modified to a fine tilth. Ap horizons, generally, are subject to a series of major compacting and loosening activities that modify their structure, appearance, and permeability throughout the cycle of seed bed preparation, sowing, crop growth, and harvest.

2. Roots and pores

The quality of the soil as a medium for plant growth is best observed through the plant itself. Observations of roots and their relation to the soil can provide indication of restrictions to root growth as well as a comparative assessment of the use of the soil by the plant species growing there. The latter observations are significant to issues such as use of available water by the plant and the effect on water balance in a catchment. For example, in Australia, there is some concern that modification of vegetation type by removal of native species and replacement with agricultural crops is increasing water accessions to rising saline water tables. Subsoil examination in these situations often reveals old root remnants in interpedal pores where there are no live roots from existing plant cover.

Soil structure, particularly macroporosity >100 μm (Gibbs and Reid, 1988), provides an indication of the available porosity for root growth and for aeration and free drainage. The significance of pores in relation to water and air movement is determined by the effective pore diameter, flow in a cylindrical pore being proportional to the radius raised to the fourth power. Hence a ten-fold increase in pore diameter (e.g., 0.1 to 1.0 mm) theoretically has a 10,000-fold effect on flow. Pores >100 μm are visible to the unaided eye and are readily assessable; size and percentage of cylindrical pores can be estimated using charts (e.g., Hodgson, 1976).

Macroporosity contributions of interpedal and packing pores are harder to estimate, but size of peds gives an indication of the frequency or closeness of these, and there is room for development of methods such as McKeague (1987), McKeague et al. (1982), and McKenzie et al. (1991).

3. Air dry consistency

Air dry consistency depends on texture, organic matter content, and clay chemistry. It is easily assessable and could be used as an indicator of soil structural condition in relation to tillage, permeability, and soil strength. Generally, air-dry strength increases with clay content; however, sands may have a modal range of air dry consistencies from loose to very weak and observed range of loose to rigid as they can become highly cemented (e.g., in a spodic horizon). Similarly, loams can exhibit hard-setting behaviour.

We might propose the ideal condition for consistency as firm when dry to friable when moist, so that the soil is sufficiently strong to resist erosion by wind or water but friable enough to allow water entry and be tilled with minimum energy across a wide moisture range. The ideal condition could be approached by managing the chemical, physical, and biological factors that control this property. So the additions of organic matter and/or bauxite residue (Ward, 1983) to a sand, coupled with minimum tillage could increase the pedality and air-dry strength of this soil. Or applying gypsum and establishing a rye grass pasture ley could decrease air-dry strength and increase friability of a sodic clay in the south-east Australian environment (Cockroft and Olssen, this volume). These management options are entirely soil specific, and the expected optimum, or at least the modal, consistency for the soil is texture dependent.

4. Aggregate stability

Visual assessments such as those cited can be combined with simple water stability tests of soil aggregates (e.g., Emerson, 1967; 1991) to monitor and improve soil quality in cultivated soils. Emerson (1991) worked with Australian Red Brown Earths (rhodoxeralfs) and a Grey Cracking Clay (ustert), but the principles could be applied to any soil. More elaborate wet-sieving methods may be applied where laboratory facilities are available, but this would not be carried out by landholders.

5. Soil colour

Soil colour is most simply represented as the product of four major influences: Fe and Mn compounds (reds, yellows, blue to grey, black), organic matter (black), salts (white), and leached siliceous material (white). These influences combine to provide the typical range of soil colours (Bigham and Ciolkosz, 1993).

Since there is a well established relationship between soil colour and principal Fe minerals present (Schwertmann, 1985; 1993), the most relevant pedogenic process that influences soil colour is gleying, redoximorphic features being good indicators of soil aeration and drainage status. As this can change through structural alteration (compaction, pans, destruction of pore continuity) and modified landscape hydrology (increased runoff, throughflow, waterlogging, groundwater discharge), soil colour has useful application in determining both the inherent soil quality and changes in soil quality. Richardson and Daniels (1993) proposed a five-fold categorisation, applying colour to assessment of hydromorphic status of pedal soils with 18–35% clay. They set these textural limits because of the strong influence that texture has on structure and therefore on the movement of air in soil; this would give rise to different manifestations of redoximorphic features in apedal soils and in soils with high clay contents. Care needs to be exercised in interpreting all redoximorphic features as contemporary. For example, James and Fenton (1993) investigated the effects of drainage on soil morphology in paired landscapes. Drainage had been installed 80 years previously, but they concluded that "based on soil color, a field soil scientist will have difficulty separating drained and undrained phases of these soils. The best criterion for separating the drained and undrained phase is land use." On the other hand, where drainage has deteriorated and waterlogging conditions develop, changes in soil colour and development of mottles, particularly in Ap and A2 horizons, can develop noticeably within a few years (Crompton, 1952).

Iron mobilisation and accumulation can also result in formation of thin Fe pans, and the development of these can be quite rapid under modified soil conditions. They have been noted to develop within five years in soils top dressed with large amounts of fibrous organic mulch (Collins and Coyle, 1980; Coyle and Collins, 1981). This is of particular relevance to the amenity horticulture industry.

6. Bulk density

Measurement of soil bulk density, particularly if the soil is sampled at field capacity, provides valuable information on total porosity as well as air-filled soil volume in pores >60 µm (or smaller, depending on soil drainage characteristics). Bulk density monitoring can easily be carried out by a landholder using a domestic oven and kitchen scales, provided that the soil volume sampled is large. Soil compaction is indicated by bulk density, but, as with air-dry consistency, this indicator is soil specific and should therefore be applied through an index of relative compaction. This is reported as a ratio of field bulk density to maximum bulk density achievable by compaction of the same soil; the latter is best standardised by using the Proctor compaction test.

IV. CONCLUSIONS

Ultimately, the land manager, as primary producer, is the person most in contact with the soil and land in its varied states and is most affected by the response of the soil to management practices. Although soil is certainly in the hands of these land managers, soil quality will only be in their hands if they have the training to recognise, and respond to, adverse soil changes. Consequently the path to soil quality assessment must include increasing the interpretive abilities of the land manager. This is most appropriate at the paddock/management unit (polypedon) scale and must entail the interaction of pedologists, agronomists, and land managers over the long term. A recent and appropriate example of such an approach is used in a soil diagnostic key for waterlogging and salinity in a south Australian catchment (Fitzpatrick et al., 1994).

The application of pedological indicators of soil quality at the paddock/management unit scale requires an appreciation of inherent soil quality, the choice of indicators appropriate to soil type and landscape position, and an understanding of the natural range of values for the selected indicator(s). The relationship between indicators, data interpretation, and scale is therefore an important and complex one that has sensitive social implications. It is not uncommon to hear negative comments about the uselessness of maps produced simply because the critics, usually the receivers of the product, and the producers, usually a government agency, are not living or thinking at the same planning scale. Increasing the capacity of land holders to implement their own monitoring, and providing means to integrate such monitoring in wider programs, are priorities that should not be ignored.

V. SUMMARY

The history of pedology, soil classification, and essential concepts supporting pedological research have been briefly reviewed and related to the paradigm of soil quality. Pedogenic processes, whether natural or accelerated by management of the soil system, have been examined for their applicability to soil quality monitoring. Broad considerations, for example, of the time scale involved and whether the processes are controllable, have been used to select a suite of indicators appropriate at the pedon and polypedon scale. Indicators have also been selected if they can be readily taken up by land managers who can be trained in their use. The interpretation of these indicators is not equally applicable to all soils and should also be made in the context of the soil landscape (catena and catchment scale) processes. The sensitivity of the indicators requires evaluation across a range of soils and production systems. A recommendation is made that soil-quality monitoring should be carried out by the landholder. To facilitate this, soil scientists and land management groups must work closely to develop appropriate programs and methodology.

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Chapter 7

EFFECTS OF SOIL REDISTRIBUTION ON SOIL QUALITY: PEDON, LANDSCAPE, AND REGIONAL SCALES

D.J. PENNOCK

I. INTRODUCTION

Soil erosion and its effects on soil productivity have long aroused interest among both the research and policy-making communities. This interest has generated a vast body of literature on erosion and its impacts, which has been ably reviewed elsewhere. The objectives of this chapter are to examine some of the recent developments in the study of soil redistribution, and to focus on their relationship to the burgeoning research on soil quality.

The connection between soil redistribution and soil quality is implicit in many definitions of soil quality. For example, Larson and Pierce (1994) define soil quality as the capacity of a soil to function both within its ecosystem boundaries (e.g., soil map unit boundaries) and in relation to the environment external to that ecosystem (particularly relative to air and water quality). The basic functions of a soil within the ecosystem are to sustain biological productivity, maintain environmental quality, and promote plant and animal health (Doran and Parkin, 1994).

Redistribution of soil may affect its ability to function by altering its chemical, biological, and physical composition at each point in the landscape where it occurs (the pedon scale). The literature of both soil science and related disciplines abounds in specific examples of the impact of redistribution on soil quality, yet few generalizations have emerged. Soil loss usually leads to a decrease in the soil's ability to function at the site of loss, but the effect of soil deposition on soil functions is less clear. The link between these redistribution-related changes in soil quality and changes in crop productivity is also elusive. Commonly, the conclusions are based on what should happen to crop yields given the severity of soil quality changes, rather than on what has been observed.

Soil redistribution also has a profound influence on the spatial pattern of soil quality indicators within the ecosystem boundary (the landscape scale). Soil redistribution can increase the range of variability within a given landscape unit. However redistribution also imposes or reinforces a distinctive landform–soil property relationship that can be used to stratify landscapes into distinctive response units or experimental units. Hence, although redistribution may increase the range of variability, it can also create or exaggerate an overall spatial order within the ecosystem.

Soil transport beyond the boundaries of the source ecosystem to adjacent ecosystems (the regional scale) is the third scale of relevance in examining the effects of soil redistribution on ecosystem function. Both the sediment itself and the chemical and biochemical components sorbed to the sediment can be significant contributors of soil-derived pollutants to aquatic ecosystems. The contribution of soil erosion to air quality (through soil transport by the wind) is important in specific regions (e.g., the dusts of north-central Africa) but tends to be a short-lived phenomenon elsewhere in the world.

A rigorous evaluation of the causes and effects of soil erosion at the three scales of relevance is critical for the development of scientifically sound land use policies. The need for reliable data upon which to base these planning strategies is starkly illustrated by two recent articles. Pimentel et al. (1995) argue that one-third of the world's arable land has been lost through erosion in the past 40 years. They estimate the cost of this loss in U.S.A. alone totals $44 billion (U.S.) per year, and suggest that these losses could be reduced to a sustainable levels with total expenditures of $8.4 billion per year. In response to this study, Crosson (1995) suggests that their estimate of arable land lost "*rests on such thin underpinnings that it cannot be taken seriously*" (p. 461). His estimate of the annual cost of erosion-induced productivity losses in U.S.A. is in the range of $500 to $600 million U.S. (1986 dollars). Clearly we cannot expect policy makers to develop sound conservation policies when we are unable to provide more authoritative data upon which to base their decisions.

II. CHANGING CONCEPTS OF SOIL REDISTRIBUTION

A. Soil loss versus soil redistribution

The term *soil redistribution* has largely replaced the term *soil loss* (or erosion) to describe the physical movement of soil components at the earth's surface (Johnson, 1988). Soil redistribution involves a continuum of processes: the initial detachment of soil from the soil mass, the movement of the detached soil away from the point of detachment (soil loss or soil erosion), and ultimately the deposition of the transported soil at the point where movement ceases. The deposition phase may occur within the field or ecosystem boundaries (in which case it is often referred to as *soil gain*) or in adjacent ecosystems (in which case the phrase *sediment deposition* is commonly used). Soil loss or erosion properly refers to only one phase of the redistribution continuum.

B. *Causes of soil erosion*

In the past, wind and water were considered the main agents of soil redistribution. Recent research on soil redistribution by tillage operations (Lindstrom et al., 1992; Govers et al., 1993, 1994, 1995; Lobb et al., 1995) has, however, challenged the view that only wind and water need to be examined as the dominant causes of soil redistribution in all landscapes. Working at research sites in agricultural landscapes of Canada and Europe, these authors have used a variety of techniques (natural and enriched ^{137}Cs redistribution, displacement of simulated clods, simulation modelling) to examine the relative importance of water and tillage erosion. Their findings strongly support the idea that tillage redistribution is the major cause of soil movement in these agricultural landscapes.

Tillage operations displace soil both upslope and downslope (depending on the direction of the operation) but because downslope displacement is greater than upslope displacement, a net downslope displacement of soil occurs. The magnitude of downslope transport depends on the type and sequence of the tillage operations, as well as the type of implements and the speed of the operation (Lobb et al., 1995).

Clearly the effect of tillage redistribution is greatest in agricultural systems characterized by highly mechanized tillage systems. Recognizing the relative importance of a given erosion process (water, wind, or tillage) in a specific region is critical for explaining the landscape-scale spatial patterns of loss and gain, as well as for evaluating the possible extra-ecosystem effects of soil redistribution.

C. *Rates of soil redistribution*

The rates of soil redistribution as reported in several sources (reviewed in Lal, 1994) are typically measured on small research plots that were designed for the development and calibration of predictive erosion models such as the Universal Soil Loss Equation (USLE) or the forthcoming Water Erosion Prediction Project (WEPP) (Lal, 1994). These observed rates usually pertain to only the soil loss part of the redistribution continuum, and hence are limited in their usefulness when examining soil redistribution as a whole.

The increasing use of ^{137}Cs as a tool to determine soil redistribution rates and patterns has provided a valuable set of observations on redistribution rates within actual agricultural landscapes (reviewed in Ritchie and McHenry, 1990). The ^{137}Cs-derived rates of soil redistribution differ widely among landscapes, regions, and agricultural systems. As a generalization, those portions of the landscape where soil loss occurs typically experience loss rates ranging from 10 to 60 Mg ha^{-1} (Table 7.1); typical average losses are on the order of 15 Mg ha^{-1} yr^{-1} (although loss at specific points may be an order of magnitude greater). Note, however, that this average

TABLE 7.1

Examples of recent values for soil loss in those portions of the landscapes that have undergone soil loss[a]

Location	Cultivation status	Rate of soil loss in eroded portions of the field ($Mg\ ha^{-1}\ yr^{-1}$)	Source
New Zealand	Cultivated	13	Basher et al., 1995
Mexico	Pasture and undisturbed forest	13	Garcia-Oliva et al., 1995
U.K., Belgium	Cultivated	10 to 20	Govers et al., 1995
Ontario, Canada	Cultivated	68 to 82	Lobb et al., 1994
Saskatchewan, Canada	Cultivated	20 to 30	Pennock et al., 1994a; Pennock and de Jong, 1990

[a]This does not represent net export of soil from the study sites.

figure does not represent soil export from the study site; in many landscapes the great majority of the eroded soil is deposited within the source landscape.

Deposition rates are considerably more variable than loss rates. Deposition is typically concentrated at a small proportion of the sampling points within a landscape (Martz and de Jong, 1987; Pennock and de Jong, 1990), and deposition rates at specific points may be very high.

For large areas of the tropical regions (including most of Africa and South America) ^{137}Cs results are not available. Lal (1995) used existing figures for sediment transport in African river systems to estimate net erosion losses in upstream areas. The great majority of the African continent experiences net soil losses of <10 (arid areas) to 50 Mg ha^{-1} yr^{-1}. The highest losses occur in the Maghreb region of northwest Africa, where losses are over 75 Mg ha^{-1} yr^{-1}.

The rates of redistribution have greater relevance for soil quality if the loss (or gains) are converted into cm of soil lost or gained. Using the bulk densities of surface Prairie soil presented in Pennock et al. (1994a), a 10 Mg ha^{-1} yr^{-1} soil loss from a soil with a bulk density of 1.07 (native prairie) translates to a loss of 0.093 cm yr^{-1}; the same rate of loss from a soil with a bulk density of 1.42 corresponds to a loss of 0.070 cm yr^{-1}. The loss per year is not dramatic, but the cumulative loss over the period of cultivation (80 years in the case of Pennock et al., 1994a) is substantial.

The sites discussed above are typical agricultural landscapes in their regions, and generally exclude representatives of high magnitude/low frequency (or catastrophic) erosion events. Sites affected by extreme events (e.g., high intensity rainfall, major wind erosion events) or locations within a field that have been selectively influenced by erosion processes (e.g., landscape positions with extensive gullying) are likely to show much higher rates of erosion.

III. SOIL REDISTRIBUTION-SOIL QUALITY INTERACTIONS

A. The pedon scale

The effects of redistribution on specific points in the landscape (hereafter called the pedon scale) begins with an understanding of the place of that pedon in the landscape. The soil and biological processes affecting a given pedon are largely driven by microclimatic differences and the redistribution of water on the soil surface and within the soil. These hydrological and microclimatic differences cause the occurrence of distinctive pedogenic regimes within the landscape, and distinct soils arise in response to these regimes (Pennock et al., 1994a).

The need to consider landscape position arises because landscape effects (differences in soil properties resulting from the position of the pedon in the landscape) can readily be confounded by erosion effects (differences in soil properties resulting from erosion among pedons at the same landscape position). Stone et al. (1985) and Daniels et al. (1989) examined the influence of landscape position on the erosion/productivity relationship for Ultisols in North Carolina. Overall they found that the most severely eroded soils in the field were usually the least productive to begin with, and that the effect of erosion on productivity decline in these landscapes had been previously overestimated by 50%. Hence, when comparing pedons it should be ensured that pedons under the same pedogenic regime in the same landscape position are being compared lest the action of erosion be confused with the position of the pedon in the landscape.

The impact of soil redistribution at the pedon scale can be broken into two types (Fig. 7.1). At sites where erosion is occurring, soil material is physically removed from the soil surface and transported elsewhere. Therefore, any layer of fixed thickness within the soil (e.g., the cultivation layer, rooting zone) incorporates an equivalent thickness of previously sub-soil material. In depositional sites only one clear effect occurs: the deposited soil buries the previous soil surface and thereby increases the thickness of the uppermost layer in the soil. Each of these effects (removal of surface soil material, incorporation of sub-soil, and deposition of soil) has distinct consequences for soil quality and will be examined separately.

1. Consequences of removing surface soil material

Erosion physically removes organic and mineral materials from the soil surface either selectively or in a mix resembling the bulk soil material. This distinction is important because of the possible occurrence of what has been termed fertility erosion (Massey et al., 1953)—where erosion selectively removes soil organic matter (SOM) and fine soil particles (clay, silt) and leaves behind a coarser lag deposit. Because the soil nutrients and exchange sites are concentrated in the SOM and clay fractions, this selective loss of material has greater effect on fertility than the bulk soil loss itself would suggest.

Inherent differences in the potential for selective removal are associated with the different erosion processes. Soil material moves as a mass under tillage erosion, and the possibility for selective movement is minimal. In the water erosion process, the possibility for selective transport can occur in interill-dominated systems, but once a

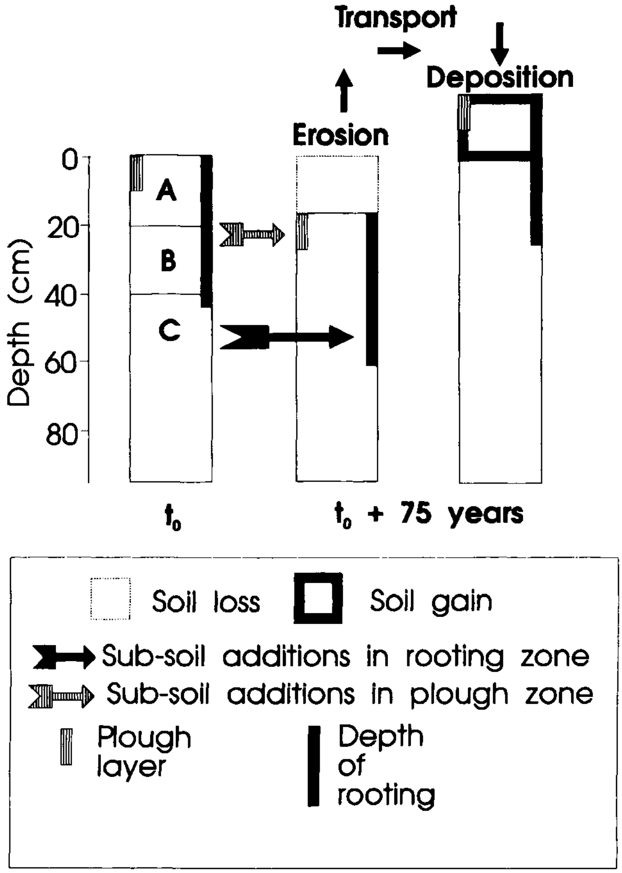

Fig. 7.1. Schematic diagram of the effects of soil loss and soil gain on individual soil pedons. The cultivation thickness is assumed to be 10 cm; the rooting zone is 45 cm. Both are characteristic of small grain production systems on the Canadian Prairies.

critical threshold is reached for a given soil type, interrill detachment and transport is not selective (Govers, 1985). In rill and gully erosion, detachment and transport is again not selective, although separation may occur during deposition. The possibility for selective transport of SOM, fine separates and fine aggregates is perhaps greatest in wind erosion, where, in extreme cases, a coarse sand–gravel lag is left behind on the soil surface (Fryrear, 1990). Hence, the concept of fertility erosion, although valid in some environments, cannot be uncritically extended to all landscapes.

The loss of SOM from the soil surface through erosion is probably the single greatest impact of soil redistribution on soil quality. The use of ^{137}Cs redistribution has allowed researchers to apportion the overall losses of SOM (or soil organic carbon, (SOC), which is commonly used as a surrogate for SOM) to redistribution and to net mineralization. For the Prairie soils of Canada, de Jong and Kachanoski (1988) suggested that approximately 50% of the observed SOC losses at the sites they examined resulted from erosion. Pennock et al. (1994a) showed that the relative contribution of erosion and mineralization differed depending on the position of the pedon within the landscape. They observed that an overall loss of 64 Mg ha^{-1} of SOC (of an original 117 Mg ha^{-1}) had occurred in shoulder positions (those with convex profile, or downslope, slope curvatures) in 80 years of cultivation; 70% of this resulted from redistribution. In footslope positions (those with concave profile curvatures), 45 Mg ha^{-1} of an original 129 Mg ha^{-1} had been lost; 18 Mg ha^{-1} (or 40%) of this loss resulted from redistribution.

The effect of these erosion-induced SOM losses on soil quality is significant. Soil organic matter and the mineral clay fraction are the two major sources of exchange sites within the soil; if SOM decreases while the clay fraction (and soil pH) remains constant, an overall diminution of CEC will occur. The effect of SOM on bulk density is also well documented; for example, Bauer and Black (1992) showed a clear inverse relationship between bulk density and SOC content for three textural groups (sandy, medium, and fine) of Haploboroll soils. The increase in bulk density in turn contributes to increases in soil strength (and possible impedance to root penetration) (Bennie, 1991); to decreases in total porosity (although perhaps not to decreases in overall gaseous diffusion through the soil; Lindstrom and McAfee, 1989, 1990); and to decreases in the infiltration capacity of the soil surface. Finally, SOM plays important and varied roles in the amount and strength of soil aggregation, and decreases in SOM will have a deleterious effect on aggregation levels in the soil (Tisdall and Oades, 1982). Kemper and Koch (1966) found a curvilinear relationship between SOC and aggregate stability; in soils with less than 2% SOC, the stability of aggregates was strongly influenced by SOC contents.

The direct impact of SOM on available water-holding capacity is more controversial. It has been argued that an erosion-induced decline in soil available water capacity (AWC) is the greatest contributor to productivity loss resulting from erosion (National Soil Erosion–Soil Productivity Research Planning Committee, 1981). Recent research differs in its conclusions on the importance of SOM to AWC: Hudson (1994) argued that increases in SOC lead to increases in AWC for the mid-west American soils he examined; however, Bauer and Black (1992) found that AWC remained constant with increasing concentrations of SOC in coarse-textured soils, whereas the AWC of medium- and fine-textured soils declined with increasing concentrations.

Bauer and Black (1994) have also attempted to directly define the effect of SOM on productivity for a loamy soil in North Dakota. For this specific soil and climate combination, they found that the contribution of 1 Mg ha^{-1} of SOM in the upper 30.5 cm of soil to inherent soil productivity was equal to 35.2 kg ha^{-1} total above-ground dry matter and 15.6 kg ha^{-1} of wheat grain. Moreover, they argued (based on their 1992 research) that the contribution of SOM to productivity is not due to its influence on AWC but is instead attributable to the nutrient contributions, especially readily mineralizable N.

The studies cited above have all focused on former grassland soils where the loss of SOM is striking, in part because of the high initial levels. In forested soils of the temperate zone, the initial levels of SOM were substantially lower than in grasslands, and the absolute loss of carbon is much less (Gregorich et al., 1995). In tropical and sub-tropical forests, Lal (1985) has argued that loss of organic materials from the soil surface is especially devastating. For example, Garcia-Oliva et al. (1995) found that 51% of the SOM, 40% of the soil potassium, 54% of the available phosphorus, and 53% of the total N are in the upper 4 cm of the tropical deciduous forest soils they examined. In these areas, where agricultural systems do not use fertilizer inputs, the possible consequences of a few centimetres of surface soil loss can be substantial.

It is common to use the term topsoil as a synonym for the organically enriched portion of the soil, but the terms are not synonymous. This is especially apparent when we examine studies that attempt to correlate decreases in the thickness of topsoil resulting from erosion with productivity decreases. Simulation models such as those of van Kooten et al. (1989) and Xu and Prato (1995) consistently demonstrate that the effect of erosion on productivity is greatest for thin soils; for example, van Kooten et al. (1989) demonstrate that yield losses are greatest when topsoil depth is less than 10 cm.

These simulation studies, however, founder on a practical point—the depth of topsoil in a cultivated field can never be less than the depth of cultivation, and the cultivated layer in a mechanized agricultural system is rarely less than 10 cm. This limitation follows from the definition of the Ap horizon in Soil Taxonomy (and other comparable classification systems): a disturbed mineral horizon, even though clearly once an E, B, or C horizon, can only be designated as an Ap. Hence, the topsoil as recognized in the field cannot be less than the cultivated layer; only in simulation studies unconstrained by reality can the topsoil be reduced to thicknesses between the depth of cultivation and 0 cm. These studies should instead use a property (such as SOC) that can vary continuously between 0 and the maximum level for that property.

2. Consequences of incorporating subsoil

For layers of fixed thickness in the soil (e.g., rooting zone, depth of cultivation), the loss of material from the soil surface results in the inclusion of subsoil material of equivalent thickness to the surface losses (Fig. 7.1). The contrast between the surface material and the incorporated subsoil material determines the severity of the effect soil loss has on soil quality and, ultimately, on soil productivity.

This statement can be illustrated with a simple example (Fig. 7.2). At the time of initial cultivation, the cultivated layer in both soils is composed entirely of A horizon material. The rooting layer of the spring wheat in the Orthic Black Chernozem (Typic Haploboroll) is unconstrained by any impeding layer; in the Orthic Black Solonetz (Typic Natriboroll), however, the contact with the salt-rich C horizon effectively limits the actual depth of rooting. Hence, even at the time of breaking, there exists a minor contrast in potential productivity between the two soils, a contrast that will be accentuated as erosion proceeds.

For both soils in this example, it is assumed that erosion occurs at a rate of 20 Mg ha^{-1} yr^{-1}, which is typical of shoulder (convex downslope) positions in the Canadian Prairies (Pennock and de Jong, 1990). This translates to a loss of

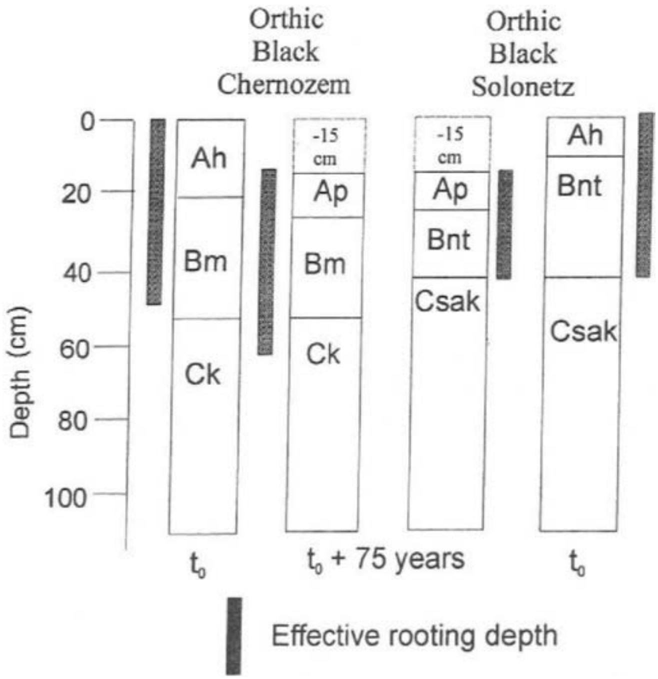

Fig. 7.2. Comparative effects of 15 cm of soil loss on a Solonetzic Order pedon and a Chernozemic Order pedon. The soil loss represents average losses from shoulder positions in the Canadian Prairies. The maximum rooting depth required is assumed to be 45 cm.

approximately 1 cm of soil every five years; hence, layers of fixed depth incorporate 1 cm of subsoil every five years to balance losses at the soil surface. For the cultivated layer, the incorporation involves mixing of 1 cm of subsoil every year into the remaining 9 cm of Ap; this results in a gradual dilution of the initial surface layer with the subsoil.

For the thick Chernozemic soil, even 75 years of erosion does not greatly alter the characteristics of the cultivation layer and the rooting zone (Fig. 7.2); the cultivated layer is still entirely composed of A horizon material and the rooting zone is still dominantly within the B horizon. A considerable net export of organic material has occurred, but the effect of these exports on crop productivity is unlikely to be major, given the lack of change within the rooting zone.

The consequences of erosion losses are much more severe for the Solonetzic soil. After 75 years of erosion, the Ap horizon is now found entirely within what was initially the Bnt horizon (although because of dilution and mixing, remnants of the former Ah are detectable); the rooting zone (assuming it is limited by the salt-rich C horizon) has been reduced by 15 cm. The decrease in the thickness of the effective rooting zone, the dense structure of the Bnt, and the deleterious sodium concentrations will all contribute to a major potential decrease in productivity; this decrease will be most apparent in dry years, during which the thin rooting layer and lack of root penetration into the peds will greatly limit water uptake by the plant.

An equivalent amount of erosion has occurred for both soils. For the Chernozemic soil, the consequences are negligible from a soil quality perspective (and probably a productivity perspective); for the Solonetzic soil, however, a great diminution of soil quality and potential productivity has occurred.

Overall, then, it is expected that soils with the greatest contrast in soil quality conditions between the surface soil and the subsoil will be most susceptible to changes in soil quality (and productivity) because of erosion; soils with a constant set of properties (or at least a gradual change in levels with depth) will show little or no effect of even substantial erosion.

The importance of the nature of the subsoil was recognized in the Productivity Index (discussed in Larson and Pierce, 1994). The index includes several subsoil properties known to limit root development, and it can be used in a range of conditions. Use of the index also allows the vulnerability of a given soil to be assessed (Larson et al., 1983; Johnson, 1988). Changes in the vulnerability of a soil through time may be plotted as a vulnerability curve, which shows the changes in the index plotted against surface soil removal and allows prediction of the effect of soil loss on productivity.

Clearly, soils that have a subsoil layer with known growth-limiting soil properties are most vulnerable to deteriorating soil quality because of surface soil loss and subsoil incorporation. As the depth to the sub-soil decreases because of surface soil loss, the interaction between the sub-soil and plants increases, and the possibility for productivity limitations also increases.

The soil layers or horizons with the greatest possible effect on productivity and the reasons for this effect are shown in Table 7.2. In several of these layers the implications for productivity are clear—in no case does soil quality increase because of the presence of a salt-rich layer or a fragipan in the plough layer or the rooting depth. In the case of plinthite, the layer itself is not limiting when found at depth in the soil; it becomes limiting, however, as soil loss brings it closer to the surface and hardening of the layer to petroplinthite begins (Eswaran et al., 1990).

The argillic horizon (which is enriched with clay relative to the overlying layer) is a problematic case. The argillic layer is commonly included as a problem subsoil in erosion studies (Larson et al., 1983); however, Stone et al. (1985) found that inclusion of the argillic horizon in the cultivated layer because of surface soil loss actually improved the overall productivity of Ultisolic soils. They attributed this productivity increase to the higher AWC of these soils (resulting from higher clay contents); in drier years the higher AWC in turn increased yields over less-eroded landscape positions.

Soils with thin organic-rich layers, such as the leaf litter of forest soils or thin A horizons in non-forested soils, are also susceptible to changes in soil quality and productivity because of surface soil loss. Overall, Lal (1985) argues that in agroecosystems dominated by tropical soils the effect of erosion can be especially critical because of the dependence of the agricultural systems on the thin, organically enriched layer.

TABLE 7.2

Horizons and associated soil orders in which the incorporation of subsoil material is likely to cause major changes in soil quality and productivity conditions

Horizon	Characteristics and constraints	Soil orders commonly associated with the horizons
Salic	High salt concentration	Aridisols, Mollisols
Fragic or Duric	Fragipan layer or layer cemented by iron, aluminum or silica; high resistance to root penetration	Spodosols, Alfisols Ultisols, Inceptisols
Natric	High sodium concentration and dense structure	Mollisols
Plinthite	High iron and aluminum oxide content; hardens upon drying; high resistance to root penetration	Oxisols, Ultisols
Argillic	Increase in clay content relative to overlying soil; increases in root penetration resistance	Alfisols, Ultisols
Oxic	High possible Al^{3+} concentrations in low pH conditions	Oxisols
Spodic	High Al^{3+} or metal levels high in low pH conditions	Spodosols

3. Consequences of soil deposition

The position in the landscape where deposition occurs differs depending on the nature of the dominant erosional process. In tillage redistribution, the location of deposition is immediately downslope of the point of initiation; the possibility of off-site transport is very low, although as Lobb et al. (1995) indicates, tillage redistribution can deliver soil to locations where it can subsequently be removed by channelized flow. Deposition of soil from the wind stream most commonly occurs immediately downwind of the source of the soil; only the finest particles can be carried for considerable distances in the windstream (Fryrear, 1990).

The greatest potential for off-site effects is associated with soil transport by the channelized flow of water. Deposition of sediment transported by flowing water occurs where the energy available for transport decreases because of a decrease in gradient, a decrease in depth (e.g., where a channel widens or becomes unconfined), or an increase in surface roughness (e.g., where the water encounters vegetation). Deposition will occur within the field if one of these conditions is met; if a channel out of the field exists, then the soil may be transported to downslope water bodies.

The off-site effects of deposition are discussed below; the on-site effects of deposition on soil quality are rarely of great consequence. A significant short-term effect is the burial of seeds or young seedlings (and hence a decrease in yield in that year). Thickening of the soil surface at the on-site point of deposition leads to the development of cumulic soils (Fig. 7.1). Pennock et al. (1994a) found that deposition of soil improved almost all the indicators of soil quality in a Boroll landscape in Saskatchewan. Again, however, generalization is difficult—if the soil delivered from upslope is of significantly lower quality than the original soil, a decrease in soil quality of the surface layer will occur.

B. *The landscape scale*

In most non-level, cultivated landscapes, the spatial pattern of soil quality indicators at the soil surface is dominantly controlled by soil redistribution. Soil erosion and soil deposition are associated with distinctive segments of the landscape; in turn, the gain (or loss) of soil causes the occurrence of a distinctive suite of soil properties in the landscape segments they are associated with.

1. *Spatial distribution*

The spatial pattern of soil redistribution in landscapes depends heavily on the nature of the dominant erosion process. Wind erosion is the only process that can remove significant amounts of soil from level landscapes, and there are examples of extremely high rates of loss from level fields (Sutherland et al., 1991). The highest losses under water erosion occur in landscape positions where channelized flow (e.g., rills, gullies) occurs in the field. The slope morphology in these positions is often concave in across-slope (or plan) curvature, with large, upslope contributing or catchment areas. Tillage erosion is highest where the profile curvature of the slope is convex (Lindstrom et al., 1992). The difference between the position of highest erosion resulting from channelized flow and tillage erosion was used by Govers et al. (1995) to assess the relative effects of the two processes in their study landscapes.

As a result of the widespread use of ^{137}Cs redistribution, a well documented spatial pattern of soil redistribution in landscapes has emerged (Pennock and de Jong, 1990; Martz and de Jong, 1987; Loughran et al., 1989; Govers et al., 1995). Slope segments with convex profile curvatures (hereafter called *shoulder elements*, Pennock et al., 1987) exhibit the highest rates of loss (Fig. 7.3c); the segments with concave profile curvatures (footslopes) exhibit net soil gain. The role of the mid-slope or backslope elements (those lacking significant plan or profile curvature) differs: under tillage erosion these elements are probably dominated by transport of soil from upslope (although infilling of small depressions in the slope occurs); in a system dominated by water erosion they can be areas of significant soil loss if sufficient water velocity or depth exists (Loughran et al., 1989).

2. *A case study*

The chronosequence of soil studied by Pennock et al. (1994a) bears out the evolution of this distinctive pattern. They examined soil redistribution using ^{137}Cs at four sites on a continuous glacial till surface in Saskatchewan, Canada: a native site and sites with 12, 22, and 80 years of cultivation (Figs. 7.3a,b,c). In the native site, the distribution of ^{137}Cs was random and showed no association with landform (Pennock et al., 1994b). After breaking of the land and 12 years of cultivation, a chaotic pattern of loss and gain occurred, again with no discernible relationship to

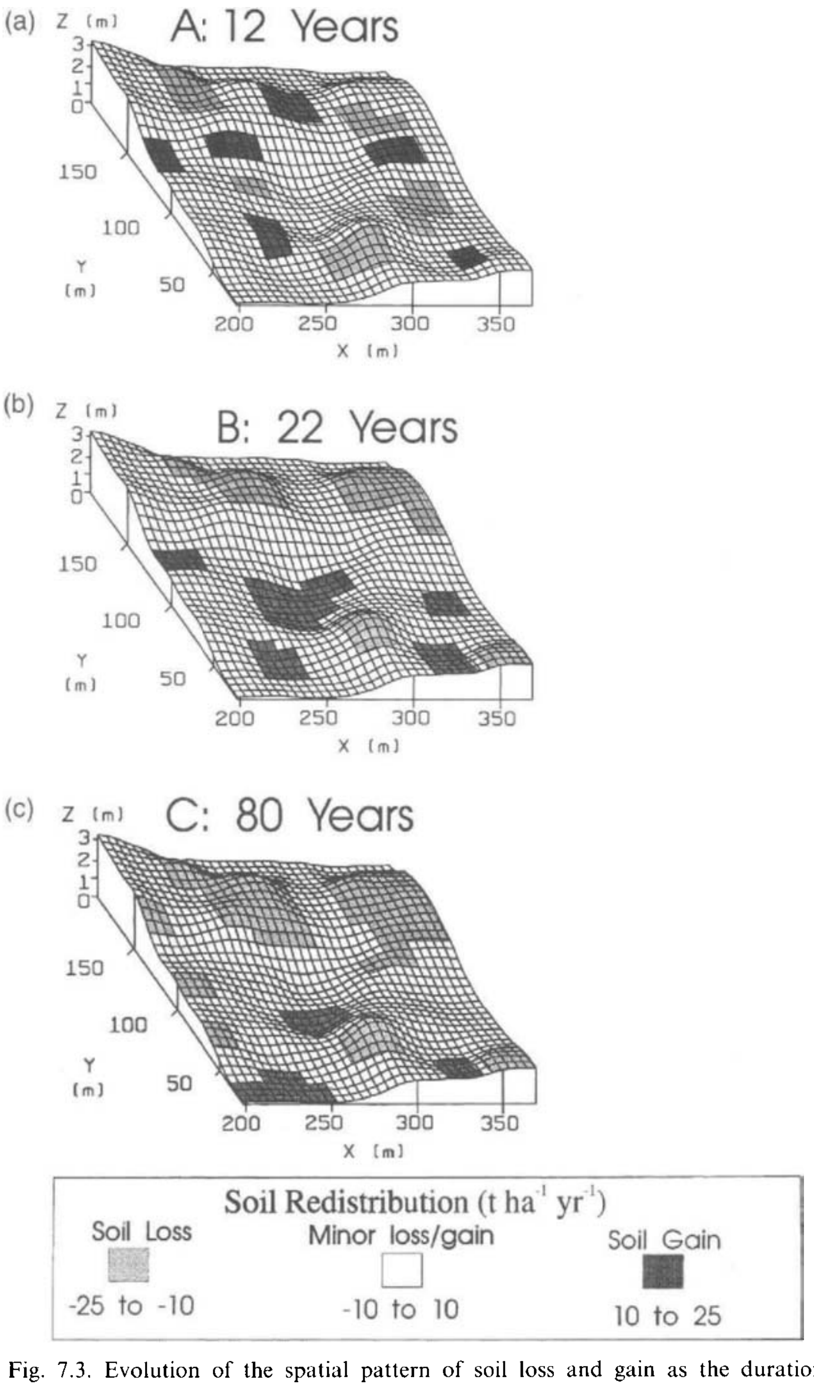

Fig. 7.3. Evolution of the spatial pattern of soil loss and gain as the duration of cultivation increases (adapted from Pennock et al., 1994). The spatial pattern shown in Figure 7.3c is characteristic of long-term (70 to 110 years) cultivated sites in the Canadian Prairies.

landform. After 22 years, however, a clear redistribution–landform relationship emerged—high soil loss in the shoulder positions and deposition in both the low-catchment area and high-catchment footslopes. After 80 years of cultivation, the soil had been scoured out of the low-catchment area footslopes and was deposited only in the high-catchment area footslope areas of the landscape.

These authors also examined the effect of soil redistribution on soil quality, and the importance of redistribution for the current pattern of soil quality indicators in their study is clear (Table 7.3). For soil pH and bulk density of the 0- to 15-cm layer, redistribution has narrowed the overall variability at the site. In both cases, the levels are approaching the values of the subsoil, indicating either the incorporation of subsoil or deposition of subsoil materials from upslope. The A-horizon differences further illustrate the problems associated with the use of topsoil discussed above: the naturally thin A horizons in the shoulders have undergone little net change in thickness, despite high ^{137}Cs-measured losses in these landscape positions. The A horizon cannot, however, drop below the cultivation depth (approximately 12 cm at this site) and hence is not a reliable indicator of absolute soil loss in these positions.

Considerable losses of soil N and SOC have occurred in the shoulder and low-catchment area footslope elements; considerable gains of these properties (and of A-horizon material generally) have occurred in the high-catchment area footslope positions. The impact of redistribution has been to reverse the spatial pattern of these properties found at the native site, where shallow-rooted wetland vegetation

TABLE 7.3

Summary of selected soil quality indicators at a native site and after 80 years of cultivation in a Boroll-dominated landscape in southern Saskatchewan, Canada (after Pennock et al., 1994a)

	Shoulder elements		Low-catchment area footslope elements		High catchment area footslopes and level depressional elements	
	Native site	80-year site	Native site	80-year site	12-year site[a]	80-year site
pH (0–15 cm)	7.4	7.9	6.6	7.6	7.2	7.7
Bulk density (g cm^{-3})	1.07	1.42	1.01	1.33	1.39	1.4
Ah/Ap thickness (cm)	14	12	22	15	11	26
Total N (Mg ha^{-1})	10.8	6.2	11.6	8.0	4.9	9.1
SOC (Mg ha^{-1})	117	53	129	84	57	113

[a]No native sites were sampled in this area, and comparisons are based on the site with 12 years of cultivation.

and anaerobic conditions originally limited biological productivity in the high-catchment area footslope areas. Hence the current pattern of soil quality indicators (at least in the surface increment of the soil) strongly reflects the action of redistribution.

C. The regional scale

In some situations the soil removed from a given landscape is exported beyond the boundaries of its source ecosystem. The major regional effects of soil redistribution are primarily associated with the water erosion process, specifically with transport in channelized flow such as in rills and gullies. In the tillage erosion process, the transport distances of any given event are small and the possibility for off-field transport are negligible. Transport within the wind stream can have episodic effects on air quality of sites downwind (Fryrear, 1990), but the possibility of medium- or long-term effects is very small.

The potential for regional effects from eroded soil transported in channelized flow depends on the sediment delivery ratio (the ratio of sediment delivered to the streams to the gross erosion occurring within the source watershed). Sediment delivery ratios differ widely depending on the size of the watershed, the soil and landform present in the watershed, the nature of the drainage network, and the vegetation cover. For example, Spomer et al. (1986) estimated a sediment delivery ratio of 60% for a small, gully-dominated watershed in Nebraska (i.e., 60% of the soil eroded in the watershed reaches the stream). For a larger watershed in Georgia with a well developed riparian forest, Lowrance et al. (1986) estimated a sediment delivery ratio of only 1%. For river systems in Africa, Lal (1995) used a sediment delivery ratio of 10% to relate stream load to erosion rates in the surrounding watersheds. Hence the potential for delivery of sediment to streams differs widely between landscapes.

Soil erosion has three main regional scale effects. First, the lag between a precipitation event and the entry of runoff water into the stream decreases as soil erosion occurs in the surrounding landscape. Increased bulk density, decreased pore space, and exposure of denser subsoils (which characterize areas of soil loss) all reduce the infiltration capacity of the eroded soils. Hence a higher proportion of the precipitation that falls is lost as runoff, rather than infiltrating the soil and slowly contributing to stream flow as lateral flow or groundwater flow. This can increase the potential for flooding in the downstream parts of the region.

The contribution of the eroded soil to sedimentation problems in the surrounding ecosystems is the second regional scale impact (McCool and Renard, 1990). Sedimentation results in the filling of roadside ditches, decreased channel flow capacity, and siltation of downstream reservoirs; eroded soil can contribute to these effects, although the specific source of sediment in fluvial systems can be difficult to assess. The increased sedimentation and turbidity resulting from soil erosion inputs can have a deleterious effect on downstream habitat for aquatic organisms.

The third regional scale impact results from the chemicals sorbed to the eroded soil, which can be released into water bodies. Materials eroded from agricultural fields may be high in basic nutrients, especially if fertilizers have been recently added. For example, Gachene (1995) found that the soil eroded from his research plots in Kenya was 247% to 936% higher in P than the source soil; the higher P levels derived from erosion of recently applied P fertilizer. The N and P inputs can contribute to eutrophication of water bodies, one of the major water quality issues in many countries (McCool and Renard, 1990). Increasingly the presence of sorbed pesticides is also a concern, although soluble forms of pesticides in runoff water appear to be a more significant contributor (ibid.).

IV. CONCLUSIONS AND PROGNOSIS

Recent developments in redistribution research, such as the use of ^{137}Cs and the increasing evidence of the importance of tillage redistribution have provided new perspectives to examine the relationship between soil quality and soil redistribution. Clearly the vulnerability of soils to soil loss differs greatly. Curiously, however, much of the research has been concentrated on soils such as the Mollisols (Chernozemic soils), whose vulnerability levels are probably quite low overall. Hopefully the wider application of relatively inexpensive techniques such as the ^{137}Cs approach will allow the soil quality soil redistribution relationship to be assessed on soils with higher vulnerability to redistribution, especially those in tropical regions of developing countries.

A dichotomy also occurs between the developed and developing worlds in terms of the prognosis for redistribution itself. In many areas of the developed world, increased adoption of reduced tillage or no-till cultivation systems has led to the increased use of herbicide-based weed control. Regardless of the cause of erosion, reductions in tillage and increases in the residue cover following field operations lower the rate of soil redistribution; in a tillage-redistribution dominated system, it may almost eliminate accelerated erosion.

The same positive prognosis does not, however, exist in many areas of the developing world. The population pressures in many developing countries may lead to increased use of marginal land and to greater use of mechanized tillage systems; both may lead to an increased acceleration of erosion. Hence, although the principles of effective erosion control are well understood and are being applied in many developed countries, the socio-economic constraints inherent to agricultural systems in many developing counties limits their application.

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Chapter 8

STANDARDISATION FOR SOIL QUALITY ATTRIBUTES

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I. INTRODUCTION

Throughout much of the twentieth century there has been very little concern about soil quality. In part this has been because we have had a resource which, whilst not infinite, has not, until recently, been considered scarce. In addition, in contrast to the other key components of the environmental system, air, and water, there are few immediate public perceptions that the quality of the soil is declining. With respect to air there has been public outcry when, as a result of pollution through the combustion of fossil fuels, the air quality is such that the population have suffered respiratory problems. Similarly with water, when the water has proved to be unfit for human consumption or the aquatic life has been killed, there has been publicly expressed concern. This "visual" evidence for changes in air and water quality has led to wide-scale public concern, and in many countries this has led to "health related" standards for air and water, on the basis of "fit-to-breathe" and "fit-to-drink" criteria. With soil there have been few such visual indicators; perhaps the most widely observed by the general public is that of soil erosion, and in some cases this has possibly led to an over-emphasis on indicators of soil quality linked to erosion. A further problem with soil is that these changes often take place gradually, and it may be difficult to observe differences except over time scales of tens of years.

For example, one soil quality indicator might be some measure of soil fertility, but how do we measure this and how do we separate the outcome of soil fertility in terms of crop yield from the normal variability that occurs from year to year because of climatic variability, the occurrence of pests and diseases, etc.? In addition, soil is a complex and varied material with exceedingly diverse and variable physical, chemical, and biological interactions, and it may be difficult to select an appropriate property to monitor soil quality.

The current concern with soil quality has arisen from two broad sources: first, the concern with land that has been contaminated by what might be broadly described as our industrial and agricultural activities, and second, the concern to view our land use activities in the long term, in particular to assess whether these activities are sustainable (leaving aside the question of how sustainability is defined). National and international discussions of the sustainable use of soils have played a key role in highlighting the need to maintain or improve soil quality. Howard (1993) recently reviewed these broad themes in the European context, discussing the different perceptions of soil quality that prevail, as well as the national and international frameworks for protecting and *monitoring* soil quality. In this chapter I shall use as illustrative material, attempts to deal with methods for the evaluation of soil quality with particular emphasis on contaminated and potentially contaminated land.

II. SOIL QUALITY FOR WHAT?

Although in recent years there has been an increased concern about the need to be able to measure and monitor soil quality, this has often been approached in a vague and unfocussed manner. If progress is to made in this area, it is essential that the question "Soil quality for what?" is answered. Pierzynski et al. (1994) considered soil quality and the causes of reduction in quality under three broad headings:

- Reduction in soil quality because of unacceptable concentrations of contaminants;
- Reduction in soil quality that limits soil function (which will probably include contamination, but also other limits to soil function, such as acidification, salinisation, and erosion);
- Soil as a source of contaminants (chiefly concerned with leaching and runoff losses of solute, suspended and particulate materials to other parts of the landscape).

It becomes apparent that a key consideration when assessing soil quality is the need to consider the actual and potential functions of the soil. Within Europe there is a suggestion that the quality of soils should be considered not with respect to a single function, but with respect to all these functions and, possibly more, its *multifunctionality* (Vegter et al., 1988). The central axiom of this approach is that the way the site is used at present should not affect its fitness for all potential kinds of use that may be possible given the natural condition of the soil. For example, the Dutch, in developing national soil protection policies, have stressed the need to preserve and protect the soil as an essential resource for the future and to avoid any human actions that reduce soil quality. Underpinning their framework of soil protection is the concept of multifunctionality, which is defined as: *The soil has to preserve the potential to perform all its possible functions. To this end the functional characteristics of the soil that are essential for the different functions must be protected* (Voorlopig Indicatief Meerjarenprogramma Boden, 1983). Although multifunctionlity has a strong appeal, relatively little thought has been given to the complexity of its meaning and almost no attention to how we might analyse the soil to assess its ability to support different functions.

Important functions of the soil in relation to its use by human beings include the possibility to build upon it, to extract groundwater, to provide raw materials, and to produce crops; ecological, hydrological, and cultural functions also have to be protected (Moen et al., 1986; Vegter et al., 1988). A similar approach was identified by Blum and Santelises (1994) when describing the concept of soil sustainability and soil resilience. They identified the need to consider ecological functions of the soil (e.g., the soil as a reactor, as a biological habitat, and as a genetic reserve) and functions related to human activity (e.g., the soil as a base for human activities, as a source of raw materials, and as part of the cultural heritage). The concept of multifunctionality recognises that soils will vary in their ability to perform different functions, but the use of this concept in the setting of standards or indicative values with respect to current soil use envisages that the use of the soil should not adversely affect the ability of the soil to perform the full range of functions. In respect of setting standards for soil clean-up, the concept of multifunctionality encapsulates the concept that there is a need to clean up the soil so that it is capable of performing a wide range of functions, not just the function with respect to the current anticipated use. The U.K. has considered the approach of multifunctionality as inappropriate in the development of soil quality standards and has adopted a more restrictive approach of judgements based on "fit for purpose", where allowable acceptable upper limits of the content of the selected soil contaminants vary depending upon the eventual use to which the site is to be put. Table 8.1 illustrates this with respect to selected metals (ICRCL, 1987).

TABLE 8.1

U.K. Interdepartmental Committee on the Redevelopment of Contaminated Land – Selected Trigger Values (ICRCL, 1987)

Hazard to health	Planned Use	Trigger Concentration (mg kg^{-1} air-dried soil)
Cadmium	Domestic gardens	3
	Playing fields, open spaces	15
Lead	Domestic gardens	500
	Playing fields, open spaces	1000
Chromium (total)	Domestic gardens	600
	Playing fields, open spaces	2000
Zinc	Any uses where plants are to be grown	130

III. CONSIDERATIONS IN THE EVALUATION OF SOIL QUALITY

A. Setting indicator values

Addressing the problem of contaminated land has provided much of the focus for recent developments in the standardisation of methods of soil analysis for the assessment of soil quality and the setting of soil quality reference or indicator values. The setting of these standards has raised many problems, both for soil scientists and for legislators. In part these problems have arisen because the soil is such a diverse material consisting of varying proportions of mineral material, organic material, water, and air (Sheppard et al., 1992). The interactions between these materials are complex, as are the interactions with added materials, and the nature of these interactions must be considered in the development of reference values. It is important to be aware that constituents of the soil, either natural or added, will be "held" to varying degrees depending upon these interrelationships Also, as illustrated below, the values reported for a particular soil property obtained by analysis and presented in reports may vary by a considerable degree depending upon the manner in which the soil is sampled, pre-treated, and analysed. It is perhaps surprising that when non-soil scientists use evaluations made against set indicator values, they may be unaware of the need to specify these important steps in the process from the soil in the field to a figure or set of figures on the sheet of paper. It is often the case that even when indicator or reference values are given for particular soil quality levels, these are difficult to use because of the poor or incomplete definition of the methods to be used in the sampling, pre-treatment, and analysis of the soil. For example, in the U.K. the Interdepartmental Committee on Redevelopment of Contaminated Land (ICRCL) of the Department of the Environment produced a set of "trigger concentrations" in 1983, revised in 1987 (ICRCL, 1987), which attempted to set "action values" for potentially contaminated land. Table 8.1 presents a subset of these values for selected metals.

Although the establishment of these reference values was an important step forward and set an important precedent in that it varied the "trigger values", depending upon a particular use rather than the much broader concept of multifunctionality, it raises many questions, because no definition was given as to whether these values were "totals" or "available" nor as to the method to be used in their analysis. The general assumption has been that the values were for "totals", but as is widely known, the "total" content of a particular metal in a soil is dependent upon the method of analysis, amongst other things.

The Netherlands produced a set of guidelines in 1983 (Moen et al., 1986; Moen, 1988; Table 8.2) that provides a contrast to the concept of "trigger concentrations" and focus upon the concept of the multifunctionality of soil and land. These guidelines or "Indicative Values for Soil Clean Up" are based on single values at three levels, perhaps more widely known as A, B, C values, and are based on the following:

A = Reference Values (levels that will not restrict the multifunctionality of the soil);

TABLE 8.2

Netherlands Indicative Values (1983) for selected metals (Moen et al., 1986)

Metal	Indicative Values		
	A	B	C
		(mg kg^{-1} dry weight)	
Chromium	100	250	800
Zinc	200	500	3000
Lead	50	150	600
Cadmium	1	5	20

B = Indicative Value for further investigation;
C = Indicative Value for clean up.

In the selection of indicators of soil quality we must seek to identify the attribute or suite of attributes that can serve as indicators of the soil's ability to perform a particular function, or that indicate changes in the soil's ability to perform that function. The identification of indicators is important in the development of a programme to monitor changes in soil quality. Once these indicators are selected, they must be monitored using appropriate standard methodologies in both the field and laboratory. Because of the availability of information, this chapter focusses on chemical analysis of soils, and indeed much of the analysis of soil quality to date has focussed upon the presentation and interpretation of the results of soil chemical analysis. Soil chemical attributes are just one part of the soil system, and there is a need to pay more attention to soil physical and biological conditions and the interrelationships between these properties that determine the quality of the soil with respect to a particular function (Papendick and Parr, 1992). In particular, soil biological conditions may provide a more dynamic indicator of soil conditions, but we must ensure that we develop methods to analyse the biological conditions that are appropriate and further that we are able to interpret this information. Because of the dynamic nature of the soil biological conditions the sampling, storage, and preparation of samples for biological analysis may have a substantial impact on the quantities derived from analysis.

B. Background values

The Dutch Indicative Values raise a very important feature of any soil quality assessment, particularly when dealing with contaminated or potentially contaminated soil. What is the natural or background level? In searching through the literature it becomes apparent that there is very little information on the natural levels of many of the "contaminants" under consideration. De Haan (1993) discusses the development of the Dutch system of reference values for various compounds, together with the manner in which these vary for heavy metals depending upon the

TABLE 8.3

Background values of selected inorganic pollutants in German soils (total concentrations mg kg^{-1}; Dinkelberg and Bachmann, 1995)

Parent material	Cd	Cu	Pb	Zn
Sands	< 0.3	13	40	51
Loess	< 0.3	25	51	89
Glacial Till	< 0.3	14	32	76
Clays	1.1	27	61	121
Basalt	0.8	71	49	168

clay and organic matter content of the soil. Although in the Netherlands, with the relatively limited range of soils and soil parent materials, it may be possible to specify with reasonable confidence the levels of background, in many other countries or regions the variability in levels that occur naturally may be considerable, but this information is often not available. The lack of such information makes an assessment of the degree or extent of contamination very difficult to evaluate. A recent study in Germany (Dinkelberg and Bachmann, 1995) has attempted to provide a summary of background levels of "total values" for selected metals in relation to broad parent material types. This summary was based on a countrywide survey. Table 8.3 provides a selection of the values identified for four metals.

These results show considerable variability across the five broad "parent material" types, but it is interesting to note that only with respect to lead for soils on clay parent materials are the "A" values of the Dutch Indicative Values exceeded. Bowie and Thornton (1985) suggest "normal" levels for the four metals listed in Table 8.3 in soils in North America: Cd < 1–2 mg kg^{-1}; Cu 2–60 mg kg^{-1}; Pb 10–150 mg kg^{-1}; Zn 25–200 mg kg^{-1}. Further to these "normal" ranges for the four metals, they highlight that values for these four metals may occur at natural sites at what would be considered contaminated levels in geochemically anomalous situations, and warn against the use of soil levels in excess of the indicator values as necessarily indicating contamination.

C. *Selecting the method of analysis*

The two sets of "indicator" or "trigger values" referred to above, and the background values have all referred to "total values" for the particular metals under consideration. There is however a potentially serious problem when dealing with total values if the method of analysis is not specified. Table 8.4 presents the results of three analyses for "total" levels of lead, zinc, and chromium in a single soil using three analytical methods. The different methods produce different "total values" for each of the metals considered, and although Method B produces the highest values for all three metals considered, the difference between the results from Method B and the other two analytical methods is relatively small for lead and zinc, but substantial for chromium. Analysis of the total levels of chromium vary by greater than 30-fold between the three methods.

TABLE 8.4

Comparison of three "total" digestion methods for contaminated soil (Duncan et al., 1995)

	Nitric-perchloric Method A	Nitric-perchloric Method B	Nitric-hydrochloric
		($mg\ kg^{-1}$)	
Lead	355	369	301
Zinc	402	406	382
Chromium	1700	6170	172

Nitric-perchloric acids Method A (0.25 g soil, 8 ml acid, 180 °C for 1 h)
Nitric-perchloric acids Method B (0.25 g soil, 8 ml acid, 180 °C for 3 h)
Nitric-hydrochloric acids (0.25 g soil, 6 ml acid, 120 °C for 2 h)

These data illustrate the importance of clearly specifying the method of analysis when presenting results, but it is perhaps more important to consider analysis in terms of the purpose to which the data are to be put. For example we might consider the following: total dissolution (e.g., geochemical prospecting); pseudo-total dissolution (aqua regia; e.g., pollution studies); selective extraction (e.g., to determine mobility). Alternatively the selection of the method of analysis might be related to the pathways under consideration. For example, we might consider the nature of the pathways and/or the relative importance of pathways. These pathways are exceptionally diverse, but might include human consumption (direct ingestion of soil or direct ingestion of plants grown in soil), damage to buildings, and influence on crop growth.

The appropriate analyses to provide information relevant for each of these pathways (and sub-pathways) are likely to be very different. This might lead us to consider the methods of analysis in relation to the processes and pathways under consideration. Important in this sequence is chemical speciation, for which we must address the *process* of identifying and quantifying the different species, forms or phases present in the soil, and the *description* of the amounts of these species, forms and phases. These species, forms, and phases can be variously defined, for example, *functionally*, such as plant-available species; *operationally*, such as aqua regia-extractable species; or as specific chemical compound/oxidation states. If we are to develop appropriate and acceptable methods of analysis, we must gather the information that will allow the correct decisions to be made.

D. Soil variability

A question which must be addressed in any assessment and measurement of soil quality is "What are the sources and magnitudes of variability?" It is imperative that all users of soil quality information are aware that variability in the final tabulated results of the analysis of a soil may be contributed from a variety of sources. In broad terms this variability might be temporal (e.g., non-systematic or random changes, regular, periodic or cyclical changes, trend changes), analytical (e.g., sampling, method, laboratory), and/or spatial (e.g., natural, human-induced).

Arnold et al. (1990) highlight the importance of temporal variability in soils and soil properties, and suggest that whilst all three broad sources of temporal variability must be considered, the focus of attention in soil quality evaluations is principally directed to longer term trends. They draw attention to the problem that it may be difficult in many circumstances to distinguish the temporal trends from the other sources of variability except where monitoring is over long time periods. When there are concerns with changes over time as a result of soil use, it may be necessary to include an element of soil monitoring in the evaluation of soil quality. Monitoring soils and soil properties over time provides the only means by which the magnitude and direction of changes in soil properties arising from the use of soils as distinct from natural or pedogenic changes may be identified (Billet, 1996). For satisfactory monitoring, given the often rapid changes in analytical procedures and changes in the precision of methods, it may be necessary to store samples which may be analysed at subsequent sampling dates.

There are now a number of well established soil monitoring networks in Europe. The Integrated Monitoring Programme, operated in north-western Europe to monitor the effects of air pollution, includes analysis of soils (Kleemola and Söderman, 1993). It specifies the temporal frequency of sampling, the spatial sample design, and the analytical procedures to be used. In France the "Observatory De La Qualité Des Sols" (Observation Network for Soil Quality) was established in 1985 to establish 100 representative sites throughout France and to sample each one-hectare site every five years (Martin, 1993). By 1993 10 sites had been established and five sites were at the planning stage. Wang et al. (this volume) describe a monitoring programme established in Canada in 1990 with seven sites, with a further 13 being added in 1993. These sites are observed to monitor soil changes under agricultural production systems based on the framework of "dynamic assessment" proposed by Larson and Pierce (1994).

The second broad source of variation, analytical variation, may be addressed through clear specification of protocols for sampling and laboratory methods, and the accreditation of laboratories and between-laboratory quality-control schemes (Griepink, 1993). In undertaking laboratory analysis it is necessary to be aware that repeated analysis of a soil sample for a particular constituent using the same analytical procedures will result in a population of measurements that are estimates of the level of the constituent in the sample. The differences between the central estimate (frequently the mean) and the "true" content is described as the *bias* of the measurement set. The distribution of the estimate is described by the *precision* of the central estimate; and if the population is normally distributed, this is described by the method standard deviation. The accuracy of an individual measurement of a contaminant in a soil sample is dependent on both bias (systematic errors) and precision (non-systematic errors). The bias of a set of analytical results may be further considered as a combination of both the intrinsic bias of the method used (method bias) and that associated with the practices in the particular laboratory in which the method was used (experimental bias). The precision of different analytical methods can vary considerably. Furthermore the precision achieved by individual laboratories using the same method may differ by a substantial amount, depending on the exact apparatus used and the skill of the analyst. Estimating the precision of the soil analysis is important, because it provides the confidence that can be attached to the conclusion as to whether the true contents of samples are above or below some threshold level of interest.

If the task is to judge whether a soil has a level of a particular constituent which is above or below a given threshold value, it is essential to have a prior estimate of the precision error associated with the chosen method of measurement. Without this information it is impossible to assign a confidence level to the allocation of samples into categories above or below the threshold value. In practice, precision is estimated by repeat analysis of soil samples and soil extracts, followed by the calculation of method standard deviation.

Spatial variability is often substantial and is frequently more difficult to deal with. It is essential, however that users of soil information are aware that soils are inherently variable in their natural context and that human activities frequently impose further patterns of variability on these natural patterns. Khan and Nortcliff (1982) investigated the spatial variability in levels of selected soil micronutrients in a relatively homogeneous soil landscape in southern England and concluded that the magnitude of spatial variability was substantially greater than variability arising in the laboratory. They further concluded that the number of samples required to achieve levels of precision in sample estimates of soil micronutrient levels within 10% of the true mean at 0.05 significance level within the landscape was daunting. Similarly Warrick and Nielsen (1980) reported investigations of the sample sizes required to obtain estimates of the physical properties of soil at the same level of precision as Khan and Nortcliff. They report that within a relatively homogenous site it was necessary to take only two samples to achieve the required precision for estimates of bulk density, but 110 and 1300 samples were required to achieve the levels of precision for per cent clay and hydraulic conductivity, respectively. Spatial variability will often include both natural and human-induced sources of variation, and it is often difficult to separate the two. When the source of human-induced variation and the process producing the variation are known (e.g., whether point or diffuse source), it may be possible to design sampling strategies to take account of this variation. For example, Ferguson and Abbachi (1993) discuss the manner in which expert judgement should be incorporated into the design of sampling strategies on contaminated and potentially contaminated sites; in particular, emphasis is given to multi-stage sampling to increase the possibility of identifying contamination hot spots, which is of particular importance if remediation to some improved soil quality is part of the aim of the investigation.

E. Environmental context

An important question that must be addressed when we are concerned with the potential environmental impact of particular soil components is how we relate the results of our analytical procedures to the environmental context in which we find the soil (Bavinck et al., 1988). We must consider whether we analyse the soil material under standard conditions (e.g., of pH and temperature) or under ambient conditions. The normal procedure in soil science is to choose standard conditions, but in taking this decision we must consider how the results are to be interpreted in the context of the ambient conditions.

IV. STANDARDISATION FOR SOIL QUALITY

A. *What is a standard?*

The evaluation of soil quality is a complex and expensive process. With regard to standardisation, it is important to recognize that there is a difference between standard methods (such as the determination of soil pH) developed by a standards body and standards (threshold or trigger values) set by governments. For example, a standards body like the International Organization of Standardisation (ISO) develops and recommends procedures or methods. A standard value may be set by a government for a threshold nitrate value considered safe for drinking water.

Analytical procedures are an important component of the evaluation of soil quality but they only constitute one step in the process: *Step 1*—Site description and identification; *Step 2*— Sampling protocol; *Step 3*—Sample storage; *Step 4*—Sample pre-treatment; *Step 5*—Analysis; *Step 6*—Interpretation; *Step 7*—Presentation of results. Although most attention has been paid to the analytical procedures, it is essential that equal attention is given to the other steps in the process, because failure to satisfactorily undertake anyone of these steps may invalidate the whole evaluation process. Vangaans et al. (1995) review some of the problems of establishing soil-quality monitoring programmes; in particular they emphasise the need to take account of background conditions at the local and regional scale.

B. *International standardisation*

In the context of the increasing concern about the quality of the soil and the absence of reliable and comparable methods of analysing this quality, the ISO established a technical committee (ISO TC 190) in 1985 to consider the development of standard analytical techniques (Hortensius and Nortcliff, 1991; Hortensius, 1993; Hortensius and Welling, 1996). This Technical Committee has a sub-committee structure that stresses the need to consider the full range of soil attributes in the assessment of soil quality, including physical and biological properties, but also emphasises the need to consider the analysis of soil quality in a broader context than laboratory analysis. The ISO TC 190 has published on the following analytical aspects: Description and Codification (SC1), Sampling (SC2), Chemical Methods (SC3), Biological Methods (SC4), Physical Methods (SC5), Radiological Methods (disbanded) (SC6), and Soil and Site Evaluation (SC7).

C. *Standardisation process*

Standardisation means the development and application of technical rules, specifications, systems, and protocols (all called standards). Standardisation derives its power and status from the consensus reached among interested parties. In addition to the standardisation of a measuring method, its validation is essential where quality and comparability of environmental measurements are concerned. Performance characteristics generally considered to be relevant are precision, limit of detection, measurement range and linearity, specificity and selectivity, and repeatability and reproducibility (between and within laboratories). The ISO Technical Committee has no role in setting limits or definitions of soil quality, which is the responsibility of national governments. An important element of the development of standard methods is inter-laboratory comparisons, which provide guidance on the reliability and accuracy of a particular procedure, and assist legislators in establishing soil quality indicators. The development of Sub-Committee 7 on Soil and Site Assessments is a move away from the development of standard analytical procedures for specific determinations; it provides guidance on the use of these standard procedures in practical situations, such as on the kind and extent of soil and site characterisation necessary for specific objectives. The task will not be to establish indicative values, but rather to provide the procedures whereby the data derived from the standardised analytical procedures may be used to provide a basis for the development of indicative values. The work programme is in the early stages of development, and an initial priority has been given to the following broad areas: assessment of excavated soil with a view to re-use, groundwater protection, and public health implications.

The Soil Quality Technical Committee has already produced a large number of standards, particularly in Sub-Committees 3 and 4, and many more are in the draft stages. Published standards are available for soil sampling and several aspects of soil chemical (C, N, P, trace elements, SO_4, CO_3, CEC), physical (water content, pore water pressure, water content in unsaturated zone), and biological (toxicity to earthworms, biodegradation of organic chemicals, inhibition of root growth) properties. A number of standards are in the final stages of preparation, such as Vocabulary—Part 1: Terms and definitions relating to the protection and pollution of soil.

D. Case study

As stated above in the process of assessment of soil quality parameters, either in the context of an initial evaluation of quality or in the context of the assessment of the success of a clean-up programme to some pre-specified criterion, it is essential that the data used to characterise the soil are appropriate for the purpose. In addition, the data should be as accurate as possible, and results from different laboratories should be comparable. Table 8.5 illustrates the variability that may arise between laboratories, between extractants, and between methods of final determination of the cadmium in a sewage amended soil from Western Europe. The laboratories in this "ring test" were well established and distributed through Western Europe.

These results emphasise the need for standard methodologies at all stages. If reference is made to the "Indicative Values for Soil Clean Up" established in the Netherlands in 1983 (Moen et al., 1986; Moen, 1988), it will be seen that in 10 of the

TABLE 8.5

Determination of the amounts of cadmium (mg kg^{-1}) in a sewage amended soil from Western Europe. Results are means (standard deviations) of five replicates. For indication the Dutch "ABC" levels of the results are given

Laboratory	Extractant	
	Acetic acid	EDTA
Flame Atomic Absorption Spectrometry		
A	18.54 (0.21) B	27.00 (0.85) C
B	17.84 (0.17) B	25.04 (0.62) C
C	19.08 (0.40) B	24.60 (0.61) C
D	24.02 (2.69) C	28.65 (2.11) C
E	18.40 (0.55) B	25.02 (0.49) C
F	17.62 (0.23) B	23.32 (0.33) C
G	16.74 (0.56) B	27.89 (0.67) C
ICP Emission		
H	18.04 (0.29) B	27.31 (0.48) C
I	20.13 (0.23) C	25.51 (1.72) C
J	21.70 (0.89) C	23.93 (2.38) C
K	18.76 (1.08) B	19.60 (0.68) B
L	17.58 (1.05) B	23.72 (0.52) C

24 means of five replicated determinations made in this study, the soil would meet the requirements of the B level, but that in the remainder the soil was categorised at the C level. The analysis involving extraction by EDTA showed a level of consistency with respect to the ABC levels. However, 11 of the 12 laboratories returned mean results indicating C levels, showing that there was still considerable variability both in the means of the replicated analyses for the two determinations and the standard deviations. This set of data reinforces the need for standardisation of all stages of the process and suggests that there is a need for clear specification of the sample preparation and analytical procedures used at every stage of the process. Failure to establish and follow such procedures may produce analytical results in which the user may have little or no confidence.

V. CONCLUSIONS

The evaluation of soil quality is an important activity, anticipated to grow in importance as we become more aware of the sensitivity of the soil to damage and the need to consider the sustainable use of soils (e.g., Royal Commission for Environmental Pollution, 1996). It is not possible to define soil quality without reference to the function of the soil, for a soil that is of good quality for one purpose may be of poorer quality for another purpose. It is important, however, in some circumstances not to consider the quality of soil with respect to one function alone but with respect to its multifunctionality, because the use of the land may change at some time in the future. Where the land is to be assessed with respect to a wide range of possible functions, appropriate methodologies must be adopted. If we are to attempt to evaluate soil quality, we must first identify soil attributes that can serve as appropriate indicators of soil quality, and secondly, select standard methods of analysis for these attributes that are appropriate to this evaluation. The presentation and interpretation of analytical procedures must be made in full awareness of the nature of the soil material, the nature of the analysis undertaken, the inherent variability in the analysis, and the relationship between the method of analysis for a particular soil attribute and the use of that attribute as an indicator of soil quality. The attribute must be fit for the purpose of indicating soil quality, and the method of analysis must be fit for the purpose of providing appropriate information about the attribute. Furthermore, analyses and the results from these analyses must not be undertaken in the laboratory in isolation; they must be considered in the context of the environment in which the soil is found.

The aim of standardisation in soil quality is the development of unambiguous methods of sample collection, preparation, and analysis to achieve uniformity in measurement and testing of soil materials. Standardisation is by nature a consensus process in which all interested parties (e.g., government, industry, consumers, and consultants) are involved in all stages. Standards are normally applied on a voluntary basis, but they may in some circumstances be incorporated in a legal framework. In either case it is essential that if all the procedures (from initial sample collection through to the final determination in the laboratory) are followed by two organisations, the results obtained should be comparable. The results presented here suggest that there is still considerable progress to be made in this respect, but that international co-operation and collaboration in this field is bringing forward the possibility of acceptable and appropriate methods of soil quality standardisation.

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Chapter 9

SOIL QUALITY CONTROL

F.J. PIERCE and D.C. GILLILAND

I. INTRODUCTION

The search is on for acceptable methods to quantify soil quality and its change in response to land management practices (Larson and Pierce, 1991, 1994; Acton and Gregorich, 1995; Hatfield and Stewart, 1993; Doran and Parkin, 1994). This chapter proposes methodology for soil quality control based on an analogy with the quality control of manufactured products using statistical quality control (SQC) concepts and procedures. It introduces the fundamental principles and concepts of SQC in an effort to provide a basis for applying these principles to the task of sustaining and improving the quality of soils. A thorough discussion of SQC is not given, because SQC procedures are described in other readily available sources (Montgomery, 1985; Ryan, 1989; ASTM, 1992). We recognize that other techniques are available to assess soil quality, including procedures to analyze spatial and temporal variation in soils (Shumway, 1988; Isaaks and Srivastava, 1989; Webster and Oliver, 1990; Mausbach and Wilding, 1991); these are discussed by Wendroth et al. (this volume).

Quality control involves both monitoring and control. Monitoring is the regular surveillance of the condition of something, whereas control means to influence or regulate. Monitoring keeps track of quality but it does not change it, whereas the main objective of SQC is to change quality. It does this by controlling the processes that determine product quality, systematically reducing variability in the characteristics that define a good-quality product so that it meets the specifications and tolerances of the design (Montgomery, 1985). Because one goal of sustainable land management should be to achieve good soil quality, it is necessary to go beyond simply monitoring and to focus on quality management. For the purpose of this discussion, soil quality means *fitness for use*, analogous to product quality in manufacturing (Pierce and Larson, 1993). The analogy between product quality and soil quality is practical, because it allows the use of an extensive set of techniques in the field of SQC (ibid.). Thus, soil quality control involves the three major components of SQC: experimental design, process monitoring and control, and continuous improvement. In terms of SQC, soil quality can be described as follows: for a given land use, soil quality will be sustained or improved if the management system is well designed relative to the intended goals (quality of design) and if the components of the system conform to specifications and tolerances that the design requires (quality of conformance) (ibid.).

II. THE CASE FOR STATISTICAL QUALITY CONTROL IN SOIL QUALITY

The use of SQC for soil quality evaluation was first proposed by Pierce and Larson (1993) as a means of developing criteria to evaluate sustainable land management. They recognized that although natural resource assessment programs, such as the National Resources Inventory and the Environmental Monitoring and Assessment Program in the U.S. (Campbell et al., 1994) and the Soil Quality Evaluation Program in Canada (Acton and Padbury, 1993), use *monitoring* as the primary assessment tool, monitoring itself does nothing to affect the condition of the target resource. They concluded that "*monitoring soil quality does not affect land (soil) quality unless this information is used to identify and implement opportunities for improvement.*"

In the field of SQC, experimental design and process control are used to build quality into the process to the point that quality assurance sampling (monitoring) of the product is minimized (Pierce and Larson, 1993). The focus shifts from monitoring to a goal of quality control or quality improvement. Thus, soil quality control must include the design of management systems that do not degrade soil quality (i.e., are inherently sustainable) and the development of process-control procedures that ensure that the processes within the management systems conform to the specifications and tolerances of that design. Soil quality is exactly analogous to SQC and is intuitively appealing. Also, SQC places control primarily in the hands of the manager and the operator, who alone can effect change in the operation of the management system. Compare SQC to traditional monitoring programs, which are generally institutional, costly, and designed for a specific purpose under a limited budget and thus limited in scope, and which produce data that are often difficult to interpret.

Our discussion focuses on the components of process control. A list of pertinent symbols and abbreviations is given in Table 9.1.

A. An overview

Statistical quality control is a method to help control processes. It uses simple control charts, produced by sampling a quality parameter over time, to determine if a specific process operates within the range of "natural variation". Natural variation is the cumulative effect of many uncontrollable causes. In SQC, a process is

TABLE 9.1

List of symbols and abbreviations used to describe statistical quality control

Symbol/Abbreviation	Description
μ	mean of distribution
σ	standard deviation of distribution
$\bar{x}_i$	sample average (for subgroup i)
s_i	sample standard deviation (for subgroup i)
$\bar{\bar{x}}$	average of sample averages
$\bar{s}$	average of sample standard deviations
c_i	count at time i
$\bar{c}$	average of counts
c_4	constant
λ	mean of Poisson distribution
SQC	Statistical Quality Control
SPC	Statistical Process Control
ARL	Average Run Length
UCL	Upper Control Limit
LCL	Lower Control Limit

considered to be "in statistical control" if it operates within the range of natural variation and is considered to be "out of control" if other variations resulting from "specific causes" are present in the process. Sample means and standard deviations of a quality control parameter are plotted over time on a control chart (Fig. 9.1). The upper control limit (UCL) and lower control limit (LCL) are set based on estimated

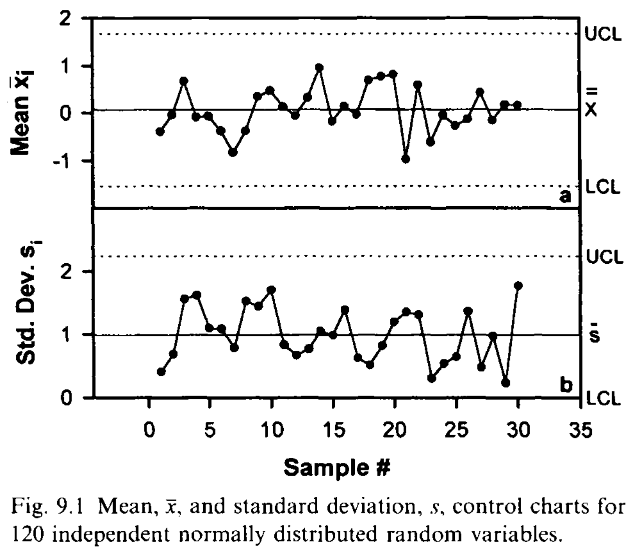

Fig. 9.1 Mean, $\bar{x}$, and standard deviation, s, control charts for 30 subgroups ($n = 4$) from a set of 120 independent normally distributed random variables.

means and variances or are known through some other means (Pierce and Larson, 1993).

Montgomery (1985) described the use of SQC in the product-manufacturing process. The design phase in product manufacturing determines the specifications and tolerances desired in the product and designs the manufacturing process that will produce that product. "*The manufacturing process transforms inputs into a finished product that has several parameters that describe its quality or fitness for use*" (Montgomery, 1985). Some of the inputs are controllable, others are not. The primary output is the product, but measures of process quality can also be output as part of the manufacturing process. Designed experiments, in conjunction with regression and time series analysis techniques, are used to identify the key variables influencing the quality characteristics of interest in the product and to determine the levels of the controllable variables that optimize the process performance. These key variables are then monitored and the process inputs adjusted as required to bring the process back to an in-control state using models that determine the nature and magnitude of the adjustments required. In dealing with a process, the stability of the process should be of primary concern; generally, little can be done to improve or manipulate a process until it is brought into control.

B. Application to soil quality

In this discussion, soil quality is denoted as **Q** and defined functionally as a set of soil attributes, q_i (Larson and Pierce, 1991). It is desirable to precisely define **Q** since the quality of two or more soils can then be compared. If **Q** is quantifiable, then the change in **Q** over time is a measure of the change in soil quality in response to management or land use. However, **Q** cannot be precisely defined, but some subset of $q_i s$ can be used to develop an estimate of **Q**, called **Q′**. If we view some subset of $q_i s$ as key characteristics or attributes of soil quality and define soil quality as fitness for use, then SQC concepts and procedures can be applied to soil quality control.

There are several aspects of SQC that apply to soil quality control. The first deals with the use of control charts to monitor each q_i and assess if soil quality is stable in response to the current management system. This statistical process-control approach indicates only that the process is either in control or not but does not in itself indicate how to bring the process back to an in-control state. If the control chart indicates that special causes of variation are present, then they should be identified and removed (if possible) to bring the process into control. Problem-solving skills, intimate knowledge of the process, and the gathering of relevant information help in identifying special causes. A second aspect of SQC applicable to soil quality control concerns the identification of key variables influencing each q_i of interest through research on the fundamental mechanisms of the process. Research involving designed experiments and computer simulation models can contribute to this basic understanding. A third aspect is the charting of the key variables and their control and adjustment to make desired changes in the q_i, which is analogous to the engineering control of a chemical process through the monitoring and adjustment of critical variables.

In the next section, we present some basics of statistical process control with examples using simulated and real data. Of course, in a given application the nature and number of variables to monitor with control charts will be determined by the nature of the application, the level of technical skills available, funding and the monitoring time frame. Our focus is on variation and whether soil quality is stable in response to management; we do not discuss experimental design and engineering control.

C. Basics of statistical process control

Soil properties and processes vary over time and space. A single measurement or set of measurements at a given time and place may carry little information about a soil property or the processes affecting it. Only by measuring a property over time and space can its nature be understood.

The charting of measurements is a standard tool in Statistical Process Control (SPC). Charts provide a history of measured aspects of the process. Charts show variation and sometimes trends and cycles as well. Processes that have *erratic* behavior are unpredictable; they are called unstable or out of control. Processes that have *regular* behavior are predictable (within limits); they are called stable or in control. Action rules applied to control charts can operationally define whether the process is in control or not.

Statistical process control encompasses a large set of techniques and ideas for dealing with data that vary in time. One technique involves plotting of means and standard deviations of a quantitative variable over time. Typically, the measurements come in subgroups that are close (local) in time, and it is the means, $\bar{x}_i$, and standard deviations, s_i, of the subgroups that are plotted. These statistics provide local estimates of the mean and standard deviation of the variable being driven by the process. The action rules that determine whether the process is in control or not are functions applied to the data in these charts.

The general model for a control chart as given by Montgomery (1985) is repeated here using terminology relative to soil quality, as follows. Let q_i be a soil quality attribute of interest, and let the mean of q_i be μ_{qi} and the standard deviation of q_i be σ_{qi}. Then the center line, the upper control limit (UCL) and the lower control limit (LCL) become:

$$\mathrm{UCL} = \mu_{qi} + k\sigma_{qi} \tag{1}$$

$$\text{Center line} = \mu_{qi}$$

$$\mathrm{LCL} = \mu_{qi} - k\sigma_{qi} \tag{2}$$

where k is the "distance" of the control limits from the center line, expressed in standard deviation units. These limits are commonly used today and may be referred to as the Shewhart limits (Deming, 1986).

Although the general form of the Shewhart control limits is as given above, the equations for the center line and the control limits depend on the type of data being generated and plotted, and on the estimates. For sample means, $\bar{x}$, and standard deviations, s, of subgroups of n measurements on a continuous variable, the Shewhart control limits for $k = 3$ are:

$\bar{x}$-chart:

$$\text{UCL} = \bar{\bar{x}} + 3\bar{s}/(c_4\sqrt{n}) \tag{3}$$

$$\text{Center line} = \bar{\bar{x}}$$

$$\text{LCL} = \bar{\bar{x}} - 3\bar{s}/(c_4\sqrt{n}) \tag{4}$$

s-chart:

$$\text{UCL} = \bar{s} + 3\sqrt{(1 - c_4^2)}(\bar{s}/c_4) \tag{5}$$

$$\text{Center line} = \bar{s}$$

$$\text{LCL} = \bar{s} - 3\sqrt{(1 - c_4^2)}(\bar{s}/c_4) \ . \tag{6}$$

where $\bar{\bar{x}}$ is the average of the subgroup means $\bar{x}_i$, $\bar{s}$ is the average of the subgroup standard deviations s_i, and c_4 is a constant that depends on the sample size n and adjusts for bias (see Ryan, 1989, Chapter 5). For $n = 4$, $c_4 = 0.9213$. The estimates $\bar{\bar{x}}$ and $\bar{s}/c_4$ are, respectively, unbiased estimates of the mean, μ, and the standard deviation, σ, of the process if the process variable is normally distributed and the subgroup measurements are random samples of size n from the distribution.

The Poisson distribution, in which the mean and the variance are equal, often describes the distribution of count data. If counts c_i are charted, the Shewhart $k = 3$ limits take the form,

c-chart:

$$\text{UCL} = \mu + 3(\sqrt{c}) \tag{7}$$

$$\text{Center line} = \bar{c}$$

$$\text{LCL} = \bar{c} - 3(\sqrt{c}) \ . \tag{8}$$

For the s-chart and c-chart, the quantities being charted (s_i and c_i) are non-negative. If the LCL is negative, then it is usually stated as zero, and the chart effectively has no lower control limit.

In some applications, it may be desirable to control within a specific level, that is, to specify standard values for the mean and standard deviation of soil quality attributes or indicators and to use these standards to establish the control charts. In this case, μ and σ are given, and the control limits of the $\bar{x}$ chart are

$$\text{UCL} = \mu + 3\sigma/\sqrt{n} \tag{9}$$

$$\text{Center line} = \mu$$

$$\text{LCL} = \mu - 3\sigma/\sqrt{n} \ . \tag{10}$$

The selection of standards is important in soil quality control, since standards in measurement and for control limits will assist in the detection of real change in soil quality.

D. *Examples of control charts*

1. Example 1: Generated data for in-control process

To illustrate the use of control charts, we generated a data set of 120 independent normally distributed random variables and placed them in 30 subgroups ($n = 4$). Envision these 30 subgroups to represent samples of a soil quality parameter, q_i, taken over a period of time. These subgroups have a mean, $\bar{\bar{x}} = 0.0646952$, and an average standard deviation, $\bar{s} = 0.9848$. The subgroup means and standard deviations were plotted in control charts (as per Fig. 9.1). The upper and lower control limits for the $\bar{x}$-chart were 1.66805 and −1.53866, respectively, and were 2.2316 and 0.0, respectively, for the s-chart.

The control charts in Figure 9.1 indicate that the process is "in control", because the variation of both the mean and standard deviation of the subgroup samples are within the range expected (within the control limits), and there are no apparent trends in the data (i.e., the variation is random and not dependent on the process).

The appearance of trends in a soil quality data set should be evaluated, as they can occur even though the data fall within the control limits and are important in exposing a change in the process or system. There are a number of tests or action rules available to identify trends in control charts. Some commonly used tests are given in Table 9.2. The data in Figure 9.1 were subjected to these tests, and neither the mean nor the standard deviation charts exhibited any detectable trends. In Example 1, the data were generated as a stable process and the action rules indicated that the process was in control.

In using control charts, there is a danger of over-analysis in the search for out-of-control indications. Assume that points 3 through 7 were examined as part of a soil quality evaluation study. It would appear from these data that both the mean and standard deviation were declining over time. In examining only these data, there would likely be a strong inclination to conclude that the process was changing in a

TABLE 9.2

Some action rules for Statistical Process Control charts

Test #	Description
1	Any point beyond the control limits
2	Runs above or below the center line longer than 8
3	Runs up or down of length 7 or greater
4	Sets of 5 consecutive points with 4 beyond 1 standard deviation in the same direction
5	Sets of 3 consecutive points with 2 beyond 2 standard deviations in the same direction

significant manner and that an adjustment in the system was needed. This conclusion would of course be incorrect, given the full data set in Figure 9.1. Any action to adjust the system would be detrimental, because tampering with a stable system will often make the system less stable (Deming, 1986). Thus it is necessary to know how measured quality variables vary over time (and space), so that real changes and special causes are detected. It is also necessary to stay with predetermined action rules rather than to selectively inspect the data and make intuitive decisions on an ad hoc basis.

2. *Example 2: Generated data for a shifted process*

To further illustrate the use of control charts, we generated a second data set of 400 normal random variables to create 100 subgroups ($n = 4$). The first 200 variables were generated from a normal distribution with a mean of 0 and a standard deviation of 1. The second 200 variables were generated from a normal distribution with a mean of 0.5 and a standard deviation of 1 to simulate a change in the system. These data are plotted in the control charts in Figure 9.2. Note that the standard deviation remains stable, as expected, since the population variance was not altered. The shift in the mean is visually evident soon after the data from the shifted distribution are encountered. However, the subgroup means fall within the control limits until sample 77 and remain within the control limits for the remaining samples. Two of the other tests in Table 9.2 for detecting trends did reveal the shift in the

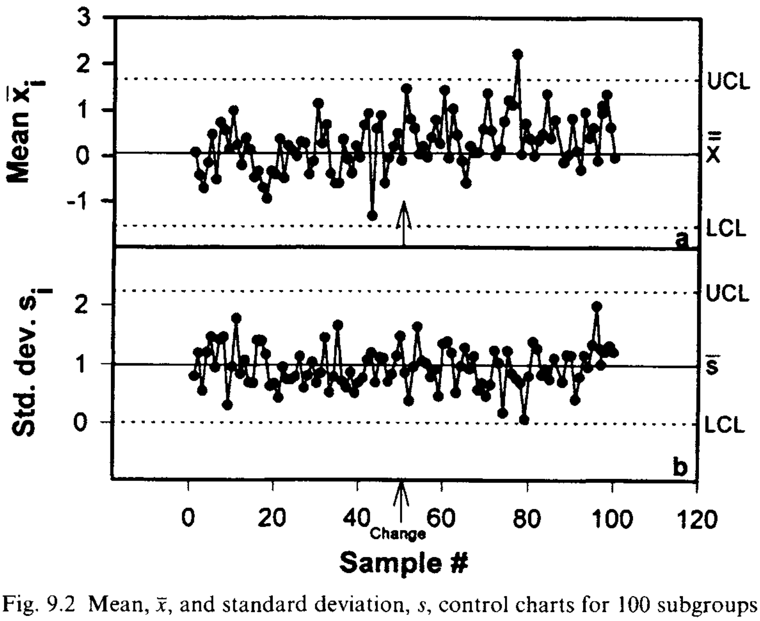

Fig. 9.2 Mean, $\bar{x}$, and standard deviation, s, control charts for 100 subgroups ($n = 4$) from a set of 400 independent normally distributed random variables, with the first 200 variables generated from a normal distribution with a mean 0 and standard deviation 1 and the second 200 variables from a normal distribution with a mean 0.5 and standard deviation 1.

mean, but not until sample 77. Thus, a shift in the process was not detected for some time after the change had occurred.

In SPC, the expected or average run length (ARL) until an action rule signals an out-of-control situation is a measure of the power of a rule to detect the change. Rules based on accumulated deviations from the center line in one direction or the other are known to have smaller ALRs than Action Rule 1 (Table 9.2) in the presence of small shifts of mean (see Ryan, 1989, Chapter 5). Of course, a shift would be detected much earlier by rules 1, 4, and 5 if the control limits were reduced. Thus, questions as to what should be the control limits relate in part to the level of change that must be detected and the cost of responding to false positives. In some cases, this change in the mean may be inconsequential and of limited importance to the quality goals of the design, while in others, a change of this magnitude is important and must be detected early. Thus, standards for soil quality evaluation are critical to both the monitoring and interpretation of soil quality parameters.

3. Example 3: Generated data for a shifted count process

Soil attributes manifest a number of variance structures in addition to the normal distribution. Count or enumeration data, such as worm counts, follow the Poisson distribution. Rates, such as hydraulic conductivity, are often best described as log-normal. Example 3 has been chosen to illustrate the *c*-chart of count data.

We generated a count data set consisting of 30 independent Poisson (mean $\lambda = 10$) variates. These data have a mean $\bar{c} = 8.76667$ and a standard deviation $= 2.96086$ and are plotted in the control chart in Figure 9.3a. The control limits were determined as UCL $= 17.6492$ and LCL $= 0$. All data points plot within the control chart, but when subjected to Rule 4 for charting (Table 9.2), a signal occurs at point 15, the end point of a set of five counts with four points beyond one standard deviation. This is a "false positive" in this example, since we know from the method of data generation that the process is "in control".

We generated a second set of count data consisting of 100 independent Poisson variates, the first 50 generated from a Poisson $\lambda = 10$ and the second 50 from Poisson $\lambda = 20$. These data are plotted in the control chart in Figure 9.3b using the mean and control limits from Figure 9.3a. For the first 50 data points, the process is "in control" by design. However, a "false positive" is present in data-point 43 since it is beyond the upper control limit. For the second 50 points, the process is "out of control" as, by design, there is a shift in the mean. The first empirical indication of the shift is found in point 53, which is beyond the upper control limit. Additionally, a number of run-tests (Table 9.2) were significant soon after the shift. Thus, the control chart clearly detected the shift in the mean soon after the shift occurred.

The operating characteristics of the rules in Table 9.2 are somewhat different for *c*-charts than for $\bar{x}$-charts. The Poisson distribution is not symmetric, and the tail probabilities are different from those of the normal distribution. For the Poisson distribution with mean $\lambda = 10$, the lower 3-sigma tail has a probability of 0.000045, the lower 2-sigma tail has a probability of 0.0103, and the lower 1-sigma tail has a probability of 0.1301. The corresponding upper tail probabilities are 0.0035, 0.0270, and 0.1355. For the normal distribution they are 0.00135, 0.0228, and 0.1587. The

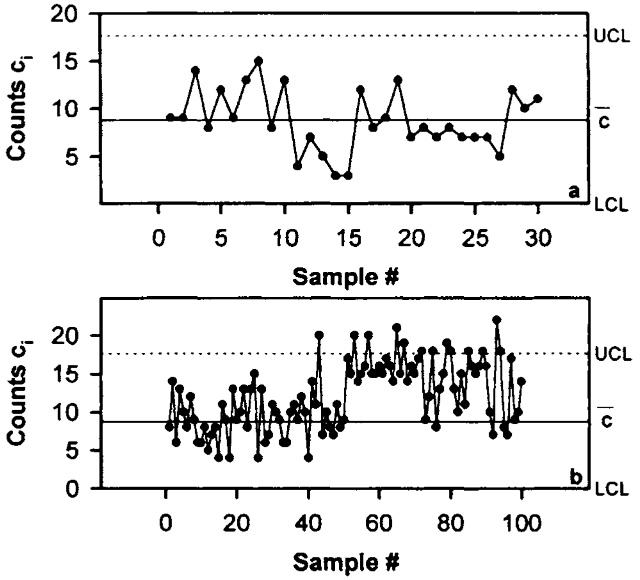

Fig. 9.3 Control charts of (a) count data for 30 independent variables from a Poisson $\lambda = 10$ and (b) for 100 independent variables, the first 50 from a Poisson $\lambda = 10$ and the second 50 from a Poisson $\lambda = 20$.

probability of falling below the mean for the Poisson ($\lambda = 10$) is 0.4579 compared to 0.5000 for the normal distribution. Ryan (1989, Chapter 5) discusses the use of probability-based control limits as alternatives to the Shewhart limits when charting.

4. Example 4: Real data for a count process

We monitored seed drop from each of six planter units on a John Deere 7200 corn planter using a standard seed monitoring unit connected to a data logger (Pierce, unpublished data). A number of variables were monitored in alternating seconds for six passes of the corn planter, including the total number of skips per second across all six planter units. A skip was defined as the failure of a seed to drop within a time tolerance corresponding to the desired seed spacing (calculated from seed drop per unit time and speed). The counts of the number of skips per second in the first three passes are plotted in Figure 9.4a (299 measurements). The average count is $\bar{c} = 5.27$ with the UCL = 12.16 and the LCL = 0. The action rules of Table 9.2 indicate out of control at time 223, which concludes a run of nine counts above the center line (6, 8, 7, 7, 7, 8, 7, 9). This is not, however, an indication of a serious departure from control for this application.

The c-chart control limits above were used as standards for the remaining three passes of the corn planter plotted in Figure 9.4b. The Shewhart 3-sigma UCL was exceeded at times 337, 341, 342, 347, 380, and 403, which concludes a run of 9, 10, 4, 10, 11. Action Rule 5 produces signals at counts 342, 343, 344, 371, 372, and 373. Thus, there is a strong indication that in the fourth pass, the corn planter was out of

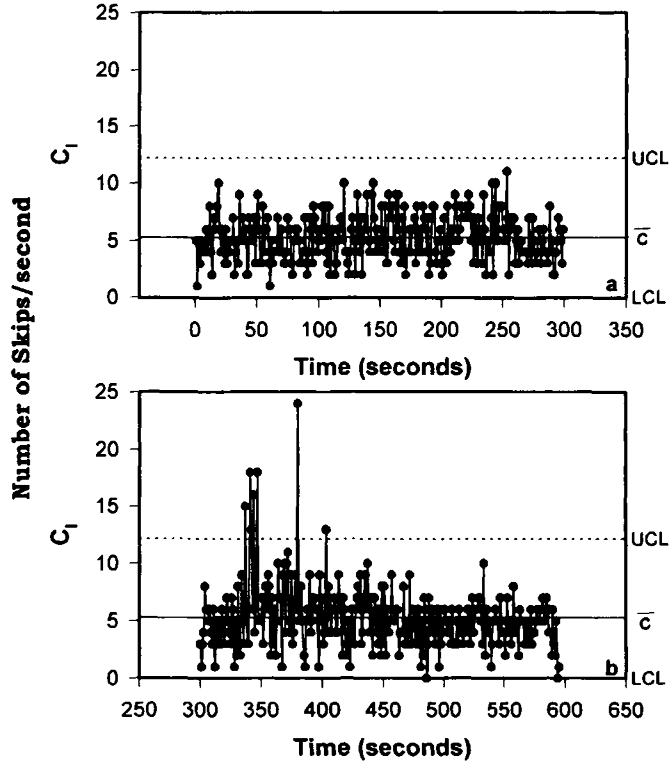

Fig. 9.4 C-control charts of (a) measurements of seed drop skips per second obtained from three passes of a six-row corn planter and (b) for a second set of three passes of the same corn planter (Pierce, unpublished data). Control limits in (a) were calculated from the count data and used in (b) to evaluate system stability.

control for the period from count 337 through count 347 and again in the period from time 371 through time 380. The signal at count 403 comes from the number of skips, 13, exceeding the UCL 12.16. These out-of-control periods should be investigated for special causes of variation within the planter units so as to adjust the planter units to an in-control condition.

5. Example 5: Real data on soil bulk density

This example illustrates the use of control charts to monitor selected soil quality parameters. Larson and Pierce (1991) proposed the use of a minimum data set (MDS) of soil attributes to monitor soil quality. The emphasis is to assess whether soil quality is changing and how it is changing (recall that monitoring does not affect soil quality). Bulk density has been proposed as one component of a soil quality MDS (Larson and Pierce, 1991; Doran and Parkin, 1994), and we will use it to illustrate the use of control charts for monitoring soil quality.

Figure 9.5 presents $\bar{x}_i$ and s_i control charts for the bulk density measurements of 20 subgroups ($n = 5$) obtained from the 0 to 8 cm depth of a fine sandy loam soil in

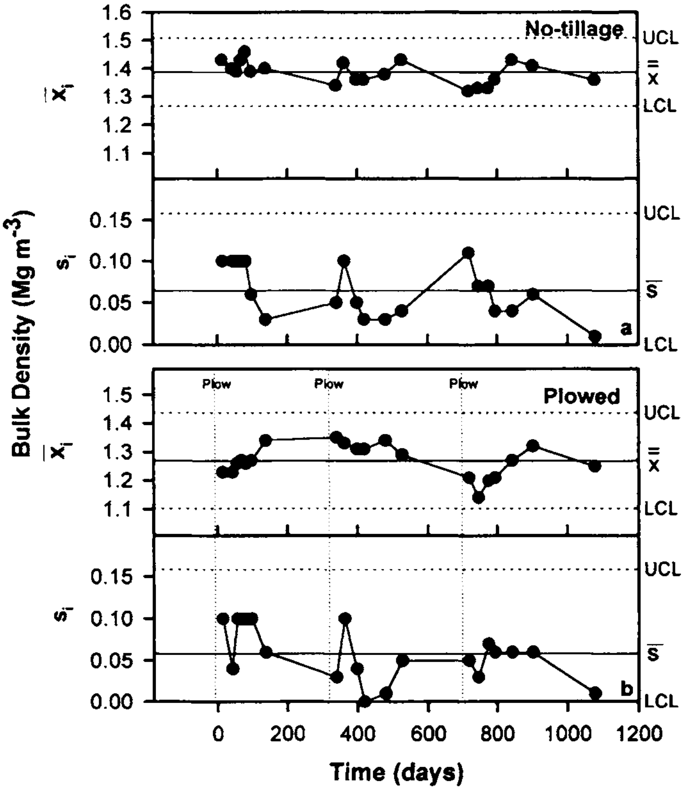

Fig. 9.5 Mean, $\bar{x}_i$, and standard deviation, s_i, control charts for surface bulk density measurements for a fine sandy loam soil in Prince Edward Island, Canada managed under (a) no-tillage and (b) moldboard plowing (from Carter, 1988, 1995).

Prince Edward Island, Canada, for no-tillage and moldboard-plowed experimental treatments (Carter, 1988; 1995). The bulk density measurements were obtained over a three-year period, 1 June, 1983 to 6 May, 1986. For no-tillage, $\bar{x}_i$ was generally stable, ranging from 1.32 to 1.46 Mg m^{-3} and a $\bar{\bar{x}} = 1.39$ Mg m^{-3}, with no apparent special causes of variation, i.e., all $\bar{x}_i$ falling within the control limits determined from the data (Fig. 9.5a). The s_i for bulk density under no-tillage was quite variable, ranging from 0.01 to 0.11 Mg m^{-3}. None of the s_i values plotted outside the control limits, but the pattern in s_i was rather cyclic, with lower s_i occurring in the fall months than in the spring and early summer. Bulk density in the plowed soil was lower ($\bar{\bar{x}} = 1.27$ Mg m^{-3}) and more variable (1.14 to 1.35 Mg m^{-3}) than under no-tillage, with broader control limits. This would be expected in response to disturbance by tillage and reconsolidation. The s_i for bulk density under plowing was similar to no-tillage in mean, range, and cyclic pattern of temporal variation. The bulk density created under each tillage system was different, but the higher bulk density under no-tillage was not necessarily indicative of soil degradation. The fact that bulk density under no-tillage was stable over time indicates that some sort of equilibrium level of bulk density is reached under no-tillage, and that this level is predictable within limits. In some cases, therefore, it may not be reasonable to apply the control limits of a soil quality parameter under one management system to limits created under another system if system dynamics are quite different, as they are in no-tillage. Interestingly, if the control limits of the plowed soil are applied to the no-tillage soil, only one point, the bulk density at day 81, plots outside the control limits. On the other hand, since the LCL under no-tillage is near the mean for plowed soil, bulk densities under plowing below the mean exceed the LCL under no-tillage.

This phenomenon appears to hold for other bulk density data sets. Control charts of the means $\bar{x}_i$ and standard deviations s_i of bulk density measurements for 15 subgroup ($n = 3$) obtained over the period from April 1987 to April 1989 from the 0 to 7.6 cm depth of a soil formed in glacial outwash in south central Michigan (data from a single experimental plot; Reinert and Pierce, unpublished data) are given in Figure 9.6. Like data from P.E.I., bulk density under no-tillage is stable

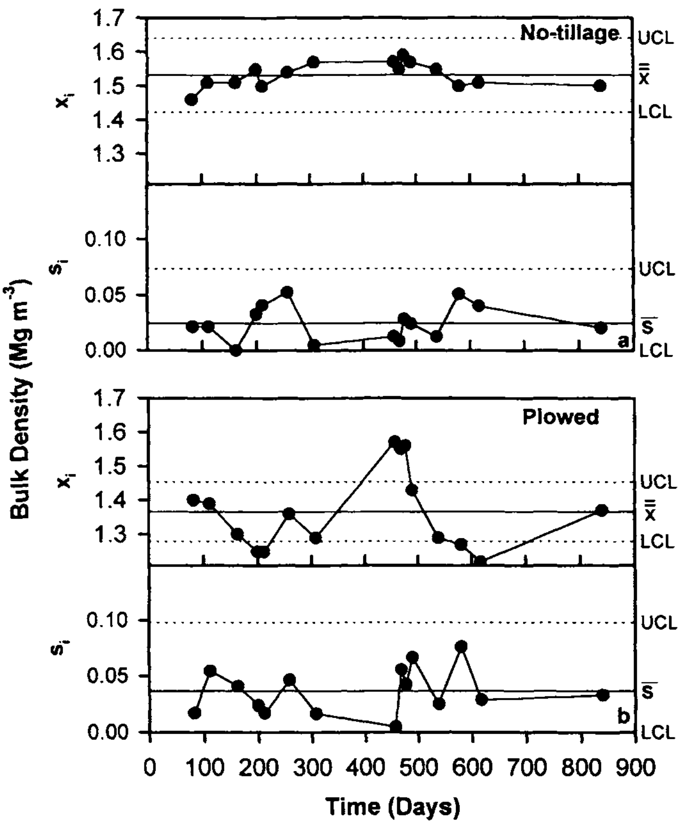

Fig. 9.6 Mean, $\bar{x}_i$, and standard deviation, s_i, control charts for soil bulk density measured from single experimental plots from a sandy loam soil in south central Michigan, U.S.A. managed under (a) no-tillage and (b) moldboard plowing (Reinert and Pierce, unpublished data).

($\bar{\bar{x}} = 1.53\,\mathrm{Mg\,m^{-3}}$), but the s_i is not as variable nor as cyclic (Fig. 9.6a). Under plowing, the bulk density is highly variable, with a lower mean value ($\bar{\bar{x}} = 1.37\,\mathrm{Mg\,m^{-3}}$) than under no-tillage (Fig. 9.6b). The s_i under plowing is similar to that under no-tillage and not subject to the wide variation as $\bar{x}_i$. There are six points outside the control limits on the $\bar{x}$-chart for bulk density under plowing (Fig. 9.6b), whereas none of the P.E.I. data (Fig. 9.5b) plot outside the control limits.

The control charts in Figures 9.5 and 9.6, however, demonstrate the value of charting soil quality variables and clearly point out the need for repeated measures to assess the dynamics of soil quality. It is important to evaluate intrinsic cycles and trends associated with the spatial and temporal variability of a property, and these can be identified with the control charts (Montgomery, 1985; Ryan, 1989). As indicated earlier, a single measurement in space and time is difficult to evaluate.

Using traditional SPC charts alone can be of questionable value. A proper analysis of bulk density and its variability over growing seasons should include estimates for cyclic components. Then charts should be made for the adjusted time series or for the separate periods of the growing seasons. Clearly, there are not enough data here to support such analyses.

III. USING STATISTICAL QUALITY CONTROL TO REGULATE SOIL QUALITY

There are two applications of SQC to soil quality. One is its use in monitoring programs where a set of soil attributes, a soil quality MDS, is charted over time. The control limits for each quality parameter would be based on the charted values or on values determined by experts or from previous experiments. The goal, for example, could be to maintain or enhance soil organic matter content at some level (in control). We envision that each parameter constituting a soil quality MDS would be charted independently, as in Figures 9.5 and 9.6, with appropriate rules (e.g., Table 9.2) or tests applied to the interpretation of each chart. Each parameter's control chart would be interpreted separately or in light of information on other parameters, because each chart provides different but possible correlated information about the state of soil quality. This procedure would allow a simple, easy-to-use, yet quantitative evaluation of the impact of a particular land use or soil management practice on soil quality. The problem is that, although these charts may detect a change in soil quality, it tells nothing about the process that created the change.

The other, perhaps more important, application of SQC lies in the process-control domain of soil quality, which is generally called engineering control (e.g., example 4). In this application, the goal is still soil quality, but the approach focuses on the process that regulates quality rather than simply monitoring a set of soil quality parameters. Consider soil organic matter, but this time think in terms of the processes that regulate organic matter content in soil. Recall that SQC involves design and process control as major components. Thus, the first matter of concern is whether or not the design of the management practice can even produce the desired level of organic matter. If the design is wrong, the desired output can never be achieved. If the system design is correct, then the important questions relate to whether operational components of the system are working within the design specifications and tolerances. If the processes that create the outcome are not in control, then the desired output may not be achieved. In SQC, if the design is correct and the manufacturing process is stable (in control), then the quality of the product is good. Thus, for a properly designed management system, the key to good soil quality is process control.

The objective of engineering control is to identify the key variables controlling the process and to chart those variables as in Figure 9.1. In the case of soil organic matter, the key variables relate to the amount and quality of organic matter returned to the soil, the rate and degree of decomposition, and the amount lost by erosion or gained by deposition. Some set of these variables would be chosen to be monitored over time and space. Further, some control limits would be selected that, if the process was in control, would achieve the desired soil organic matter content. If the control charts for these key variables are out of control, then the process would be adjusted according to design criteria.

In another example, the goal might be to achieve a specific amount of crop cover or crop residue cover on the soil surface after tillage. The key control variables would likely be associated with the tillage tool components that regulate how much crop residue is buried (e.g., operating depth, speed, implement angle or number). As posed by Pierce and Larson (1993), simply informing the farmer after planting that crop residue cover on the field doesn't meet erosion control guidelines is too late to affect the process that created that condition. Adequate residue cover could be achieved if the key variables were charted and adjustments were made to bring the process back into control during the tillage operation (as in the seeding rate, Example 4). An important aspect of this approach is that it places the responsibility for soil quality in the hands of the land manager, which is the only way to really cause a change in soil quality.

It may seem that system design and engineering control are not practical in agricultural management systems. However, emerging technologies associated with site-specific management and precision agriculture are, in fact, capable of achieving this level of design and control. Technologies include very accurate location control (e.g., global positioning systems, GPS), variable rate application equipment, and sensors for both control and performance evaluation (e.g., yield sensors). These technologies are now readily available (Larson and Robert, 1991; Schueller, 1992; Robert et al., 1993, 1995). The potential for real-time control is rapidly emerging and may be the best means of achieving good soil quality. While monitoring will continue to be a focus in soil quality assessment, it is soil quality control that will affect the soil resource base.

IV. CONCLUSIONS

The analogy of soil quality control to quality control in product manufacturing is presented as a model to achieve soil quality. The analogy is not perfect, but the use of SQC is intuitively appealing given the goals of sustained or improved soil quality. This chapter merely introduces the topic and the concepts involve in SQC. As SQC is applied to soil quality, considerable refinement of the concepts and procedures in SQC to this application will emerge.

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Chapter 10

PEDOTRANSFER FUNCTIONS TO EVALUATE SOIL QUALITY

J.H.M. WÖSTEN

I. INTRODUCTION

Intensive agricultural and industrial activities in many regions increasingly cause the quality of our soils and waters to deteriorate. The use of fertilizers, pesticides, and inorganic and organic chemicals has already caused considerable environmental damage (Eijsackers and Hamers, 1993).

In order to partly control and ultimately rectify this damage, scientists have developed increasingly complex computer models to simulate water and solute movement in the unsaturated zone of the earth's crust. These models have now become indispensable in research directed towards quantifying and integrating the most important physical, chemical, and biological processes taking place in the unsaturated zone of agricultural soils (Addiscott and Wagenet, 1985). Models ranging from very simple to highly complex are being used in a wide variety of studies, such as land evaluation, water management, soil protection, predictive studies, and global climate change (e.g., Boesten and Van der Linden, 1991; Kabat et al., 1992; Teng and Penning de Vries, 1992).

Because environmental changes are not restricted by country borders, there is a general consensus that they should be studied in a global context. As a consequence, international organizations in the field of agriculture and environment are funding projects to develop a more sustainable use of our soils and waters. The use of models for research and management has shown that many input data have to be quantified in order to make reliable predictions. At the same time these data are usually fragmented, of different degree of detail, of varying reliability, and are held in different institutes scattered over the world. It is therefore important to develop methods that overcome these limitations.

Nowadays, in various fields of model application, such as hydrology, environmental risk analysis, or assessment of global climate change, the lack of relevant characteristics (particularly with respect to soil hydraulics) is considered a major obstacle to progress. As our ability to numerically simulate complicated flow and transport systems increases, reliability of model predictions may well depend on the degree of detail with which we can estimate model input data such as climate, rooting patterns, water table fluctuations, soil chemical characteristics and, above all, soil hydraulic characteristics (van Genuchten and Leij, 1992). The latter characteristics (water retention and hydraulic conductivity curves) are key characteristics in this respect. The problem is also aggravated by the awareness of the significance of effects of temporal and spatial variability in hydraulic characteristics on model results, which means that many more samples are needed than previously thought to properly characterize a given field (Warrick and Myers, 1987). Besides variation in hydraulic characteristics from one location to the other, there is also an important temporal variability in hydraulic characteristics. For example, cultivation practices, shrink and swell phenomena, and soil crusting cause hydraulic characteristics to vary with time.

Input data required for simulations can be obtained from direct measurements using different laboratory and field techniques (e.g., Klute, 1986; Page et al., 1982). However, the problem is that most of these techniques are relatively time consuming and costly. At the same time, good predictions instead of direct measurements of input data may be accurate enough for many applications. Therefore, the need to make new measurements has to be critically evaluated considering both the desired accuracy of the input data and the available financial resources to measure them. It is necessary to evaluate whether there exists a balance in levels of detail of the different input data as well as in the level of detail of the applied simulation model, because studies in which a good balance is maintained are the most promising ones.

Therefore, it is beneficial to analyse existing data bases in a way that allows input data to be predicted from existing measured soil data if their direct measurement is difficult. An example is the prediction of soil hydraulic characteristics from data recorded in soil surveys, such as percentages of clay, silt, and organic matter content. This latter procedure is called an indirect method or pedotransfer function (PTF) approach (e.g., Bouma and Van Lanen, 1987; Larson and Pierce, 1991; Hamblin, 1991). Larson and Pierce (1991) present a limited list of available PTFs and discuss their role in providing a minimum set of soil data required to assess soil quality.

This chapter focuses on the description of PTFs, with emphasis on functions to predict soil hydraulic characteristics, using data mainly from the Netherlands. The practical use of the PTF approach is demonstrated in a case study on functional aspects of soil behaviour. PTFs used in soil chemical and soil fertility studies are briefly examined, as is the quantification of uncertainty involved in using PTFs and multicollinearity in regression models.

II. USE OF PEDOTRANSFER FUNCTIONS FOR SOIL HYDRAULIC CHARACTERISTICS

Darcy's law states that water flux in porous media equals hydraulic conductivity times gradient of soil water potential. Combination of Darcy's law with the expression for conservation of mass yields the Richards (1931) partial differential equation for water flow in unsaturated soil:

$$\frac{\delta\theta}{\delta t} = \frac{\delta}{\delta z}\left[K(h)\left(\frac{\delta h}{\delta z} - 1\right)\right] \tag{1}$$

where h is the soil water pressure head, θ is the volumetric water content, K is the hydraulic conductivity, t is time, and z is soil depth.

To solve this equation, information is needed on the h (θ) water retention and $K(\theta)$ or $K(h)$ conductivity relationships. Because of their importance, much work has been done in order to establish accurate values for these relationships. Roughly speaking, investigations to determine the relationships can be divided into a direct measurement approach and an indirect prediction approach. In the following sections, the different approaches are discussed with an emphasis on the indirect PTF approach.

A. Direct methods for measuring hydraulic characteristics

Smith and Mullins (1991) gave an overview of laboratory and field techniques for direct measurement of the soil hydraulic characteristics. Many direct measurement techniques require steady state conditions and are based on direct approximations of Darcy's law. Many transient methods are also widely used, such as the hot air, sorptivity, disc permeameter, and instantaneous profile methods. Although these methods are relatively simple in concept, they have the disadvantage that they are often time consuming (waiting for steady state), and that a number of boundary conditions must be obeyed.

A variation on direct measurement techniques is the inverse modelling approach, in which a flow experiment in the laboratory is simulated by estimating the hydraulic characteristics in such a way that differences between properties, such as observed and calculated water contents, pressure heads and/or flow rates, are minimized (Dane and Hruska, 1983; Kool et al., 1987). In this approach, additional information is often required to ensure uniqueness of the predicted characteristics. Feddes et al. (1993) demonstrated that inverse modelling is also applicable on large scales when remote sensing is used to determine areal evaporation and surface moisture. Care should be taken in the latter approach that not all of the uncertainty is lumped in the determination of the soil hydraulic characteristics.

Developments in electronic and computer technologies will probably improve measurement procedures and result in computer-driven, stand-alone devices. Despite these efforts, the fact remains that at the present time direct measurement of hydraulic characteristics is notoriously difficult and costly, especially for undisturbed field soils (van Genuchten and Leij, 1992).

B. *Indirect methods for predicting hydraulic characteristics*

Although much attention has been paid to the development of direct methods for measurement of hydraulic characteristics, relatively little attention has been paid to the development of indirect methods to predict hydraulic characteristics from more easily measured data. The latter type of data (e.g., clay, loam and organic matter percentages or data on particle size distribution or water retention) might be routinely recorded in soil surveys. These indirect methods have a clear advantage over direct methods because they are far less costly and are generally more convenient to use. A possible disadvantage is that indirect methods offer a prediction and not a measurement of the hydraulic characteristics.

In numerous cases practical applications do not require very accurate hydraulic characteristics, and therefore predictions made with these indirect methods may be sufficient. When using indirect methods, two different approaches can be used. In the first approach, the more difficult to measure characteristic, hydraulic conductivity, is predicted from the more easily measured characteristic, water retention. In the second approach, hydraulic characteristics are predicted from data recorded in soil surveys where no measured hydraulic characteristics are involved. It is important to note that no indirect methods exist without direct methods to measure the hydraulic characteristics. Only direct measurements will create a data base from which indirect methods can be derived to predict hydraulic characteristics. Therefore, development of indirect methods does not imply that continued research towards improved direct methods is obsolete.

1. *Predicting hydraulic conductivity from measured water retention*

In this approach the measured water retention curve is described with an analytical function. Based on this, a pore size distribution is derived, which in turn is used to predict a hydraulic conductivity function assuming water flow through cylindrical soil pores. However, this latter assumption is highly simplified, because soils do not consist of bundles of smooth, interacting or non-interacting parallel cylindrical pores. In reality many soils shrink and swell, are structured, and have macropores, which means that pores in soil are irregular in size and in shape. Therefore, approximations of the hydraulic conductivity function with a pore size distribution model should be reviewed critically.

Vereecken (1992) and van Genuchten and Nielsen (1985) give an inventory of proposed purely empirical or semi-empirical equations for the description of water retention functions. Among them the equations proposed by Brooks and Corey (1964), Visser (1968), Campbell (1974), van Genuchten (1980), and Russo (1988) can be used relatively easily to arrive at a pore size distribution from which a hydraulic conductivity function can be predicted. Mualem (1992) showed that there are a number of methods for predicting unsaturated hydraulic conductivity from measured water retention data. Among these methods, approaches related to the theories of Childs and Collis-George (1950), Burdine (1953), and Mualem (1976) are most widely used.

Because this approach makes use of the necessarily simplified concept of cylindrical pore sizes, the resulting hydraulic conductivity functions must be validated by comparing large numbers of conductivity measurements. Due to the assumptions that have to be made, it is likely that this approach is more promising for sandy soils than for clayey and loamy soils.

2. Predicting hydraulic characteristics from soil survey data

When no measured hydraulic characteristics are available, the water retention characteristic can be predicted from the particle size distribution, bulk density, and particle density. Bloemen (1980), Arya and Paris (1981), and Arya and Dierolf (1992) presented models to do this. This type of model first translates the particle size distribution into an equivalent pore size distribution model, which in turn is related to a distribution of water contents and associated pressure heads. The model assumes spherical particles and cylindrical pores. In the model, use is made of similarity in shape of the water retention curve and the cumulative particle size distribution. The Arya–Paris model has been modified and extended by different investigators (e.g., Haverkamp and Parlange, 1986; Tyler and Wheatcraft, 1988). For sandy soils it predicted water retention functions that compared very well with measured data. However, agreement between predicted and measured data for loamy and clayey soils was less successful. In the same way predicting hydraulic conductivity from measured water retention is a problem, as it also requires a soil pore network model that adequately represents the complex pore geometry in real soils.

The above-mentioned theoretical models predict hydraulic characteristics on the basis of physical principles. Alternatively, purely empirical regression models can also be developed. In the latter case no theoretical background is provided, but emphasis is on models that give a sufficiently accurate description of, for example, the soil hydraulic characteristic. For the latter group of empirical regression models, the term PedoTransfer Functions (PTF) has been introduced by Bouma and Van Lanen (1987). They defined PTFs as functions that relate different soil characteristics and properties to one another or to land qualities. PTFs are functional relationships (Larson and Pierce, 1991) that transfer available soil properties (e.g., texture, structure, and organic matter content) into missing soil properties (e.g., soil hydraulic and soil chemical characteristics). In this respect, Hamblin (1991) used the term *surrogates* for available soil data from which missing data are derived.

A distinction is made between class PTFs (using a horizon designation or a soil type as regressed variables) and continuous PTFs (using soil properties such as clay and organic matter percentage as regressed variables). Continuous PTFs may be used to predict either specific points of interest on the θ–h–K relationships or parameters in a model describing the complete θ–h–K relationships. Class and continuous PTFs will be discussed further in the following section illustrating with the approach used in the Netherlands as an example.

III. ESTABLISHING TWO TYPES OF PEDOTRANSFER FUNCTIONS

Over the years and in a number of different research projects, soil water retention and hydraulic conductivity curves have been measured for a large number of soils in the Netherlands. As a set, the curves form a unique data base covering a broad spectrum of soils. Hydraulic conductivities are measured using a combination of the following five methods:

(i) The column method for vertical saturated hydraulic conductivity, K_s;
(ii) The crust test for unsaturated conductivities when the pressure head, h, is between 0 and −50 cm;
(iii) The sorptivity method for conductivities of coarse-textured soils when $h \leq -50$ cm;
(iv) The hot air method for conductivities of medium- and fine-textured soils when $h \leq -50$ cm;
(v) The evaporation method for hydraulic conductivities when h is between 0 and −800 cm.

Soil water retention curves are obtained by slow evaporation of wet, undisturbed samples in the laboratory. Pressure heads are measured periodically with transducer–tensiometers, and subsamples are taken at the same time to determine water contents. Alternatively, water contents are determined by weighing the total sample at each pressure head. Both methods yield points relating h to the water content θ. Water contents corresponding with pressure heads lower than −800 cm are obtained by conventional methods using air pressure.

Details on the applied measurement techniques are given by Stolte et al. (1992) and Wopereis et al. (1994). The effort to measure hydraulic characteristics for a variety of soils resulted in a data base comprising 620 measured characteristics (Wösten et al., 1994).

A. Class pedotransfer functions

To establish class PTFs, soils for which hydraulic characteristics have been measured are classified according to soil texture and type of horizon as either topsoil (A horizon) or subsoil (B and C horizons). In the classification procedure, 18 different texture classes are distinguished for both topsoils and subsoils. Figure 10.1 shows the 67 individually measured hydraulic characteristics for a topsoil horizon with a silt-loam texture, having between 85 and 100% clay + silt (i.e., percent < 50 μm). After grouping the measurements in the appropriate texture class, geometric means of the characteristics are calculated for each texture class. Geometric, rather than arithmetic means of the characteristics are calculated because K and θ values at different h values are log normally rather than normally distributed. Because average hydraulic characteristics for a texture class are presented, they are referred to as class PTFs. Figure 10.2 shows calculated geometric means of hydraulic characteristics for a topsoil horizon with a silt-loam texture, having between 85 and 100% clay + silt. Because relatively wide ranges in

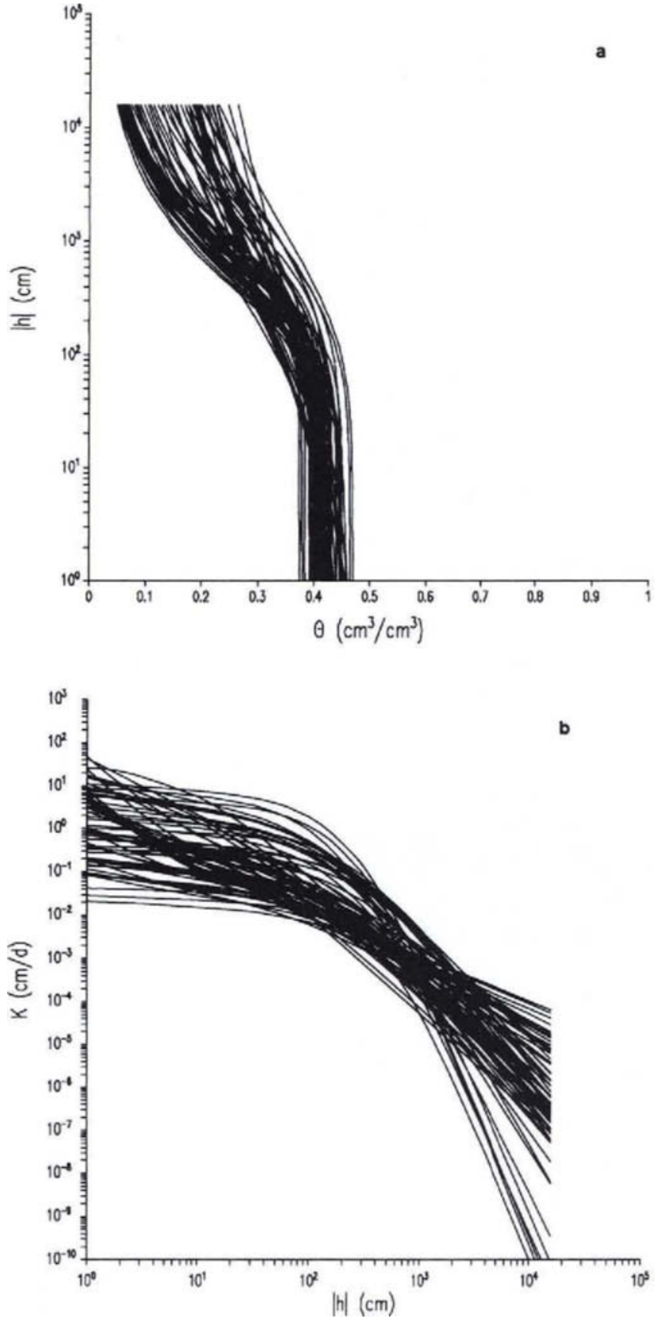

Fig. 10.1. Individually measured water retention (a) and hydraulic conductivity (b) characteristics for a topsoil horizon with a silt-loam texture, having between 85 and 100% clay + silt (after Wösten et al., 1994).

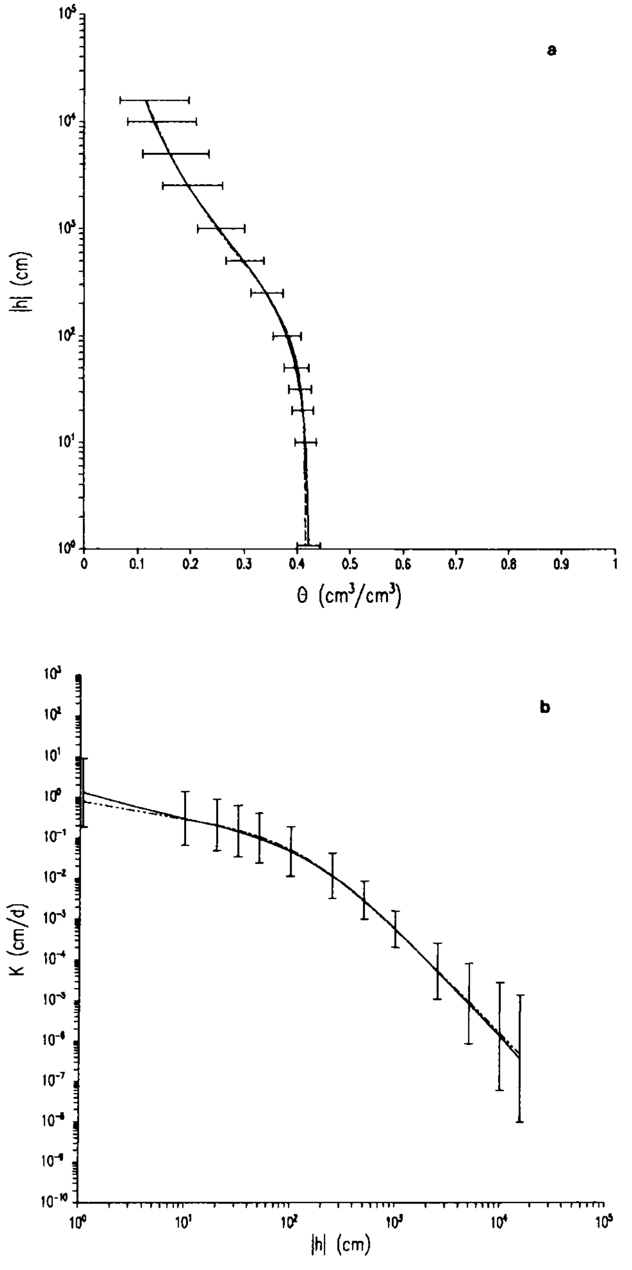

Fig. 10.2. Geometrically averaged water retention (a) and hydraulic conductivity (b) characteristics calculated on the basis of 67 individually measured characteristics for a topsoil horizon with a silt-loam texture, having between 85 and 100% clay + silt (after Wösten et al., 1994). The solid lines indicate the geometrically averaged characteristics, the dotted lines indicate the fits of analytical equations on the averaged characteristics, and the bars indicate the mean values plus and minus one standard deviation.

texture of the distinguished texture classes, individual characteristics within each class show considerable variability (Fig. 10.1), which is also reflected in the length of the bars indicating the mean values plus and minus one standard deviation (Fig. 10.2).

The resulting data set with class PTFs for major soil groups in the Netherlands has been successfully used in many studies where soil physical information at regional and national scale is required (Wösten et al., 1990). Because the PTFs present an average curve for a soil group, they should not be misused to predict hydraulic characteristics for a specific location. When more data become available, the class PTFs can easily be updated. Clapp and Hornberger (1978) presented class PTFs for soil groups distinguished in the U.S.A.; de Jong (1982) presented them for soil groups in Canada; and Mualem (1976) presented them for a wide variety of soils collected at different locations all over the world.

Calculated average characteristics are described with analytical equations to facilitate their efficient use in simulation models. A wide range of different equations is available for the parameterization of measured hydraulic functions (Vereecken, 1992). In this study volumetric soil water content, θ, and hydraulic conductivity, K, as functions of pressure head, h, are described with the following equations (van Genuchten, 1980):

$$\theta(h) = \theta_r + \frac{\theta_s - \theta_r}{(1 + |\alpha h|^n)^{1-1/n}} \tag{2}$$

$$K(h) = K_s \frac{\left((1 + |\alpha h|^n)^{1-1/n} - |\alpha h|^{n-1}\right)^2}{(1 + |\alpha h|^n)^{(1-1/n)(l+2)}} \tag{3}$$

In these equations the subscripts r and s refer to residual and saturated values and α, and n and l are parameters that determine the shape of the curve. The residual water content θ_r refers to the water content where the gradient $d\theta/dh$ becomes zero ($h \to -\infty$). In practice θ_r is the water content at some large negative value of soil water pressure head. The parameter α (cm^{-1}) approximately equals the inverse of the pressure head at the inflection point where $d\theta/dh$ has its maximum value. The dimensionless parameter n determines the rate at which the S-shaped retention curve turns towards the ordinate for large negative values of h, thus reflecting the steepness of the curve. The dimensionless parameter l determines the slope of the hydraulic conductivity curve in the range of more negative values of h. Although l is presumably a soil-specific parameter, Mualem (1976) concluded from an analysis of 45 soil hydraulic data sets that l should be, on average, about 0.5. In this study l is not fixed but is considered one of the experimental unknowns. As follows from the equations, the parameter θ_r affects only the shape of the retention curve while leaving the conductivity function unaffected. The parameter l, on the other hand, affects the hydraulic conductivity only and leaves the retention curve unchanged. The flexibility of the equations in generating different shapes of $\theta(h)$ and $K(h)$ relationships has been demonstrated by a number of researchers (e.g., Hopmans and Overmars, 1986; Wösten and van Genuchten, 1988).

The nonlinear least-squares optimization program RETC (van Genuchten et al., 1991) is used to predict the unknown parameters (θ_r, θ_s, K_s, α, l, and n) in both equations simultaneously from measured soil water retention and hydraulic conductivity data. In the optimization procedure the sums of squares of differences between measured and predicted water contents and between measured and predicted hydraulic conductivities are minimized. The RETC program is used for parameterization of class PTFs for the different texture classes distinguished in the Netherlands. The dotted lines in Figure 10.2 show the parameterization of hydraulic characteristics for the topsoil horizon in the texture class silt loam, having between 85 and 100% clay + silt. In this procedure all of the unknown parameters, such as θ_r, θ_s, K_s, α, l, and n, are optimized. The results of this optimization procedure for all texture classes distinguished in the Netherlands are presented by Wösten et al. (1994). In total 36 texture classes (18 topsoils and 18 subsoils) were distinguished. Geometric means of hydraulic characteristics were calculated for 30 of these 36 texture classes. For the remaining six texture classes, the number of individually measured hydraulic characteristics was insufficient to justify calculation of geometric means.

B. Continuous pedotransfer functions

Continuous PTFs that predict specific points of interest on the θ–h–K relationships have been developed by several researchers (Gupta and Larson, 1979; Poelman and Van Egmond, 1979; Rawls et al., 1982; Ahuja et al., 1985). These functions often have the following form: water content θ (at, for example, $h = -100\,\text{cm}$) $= b_0 + b_1 * C + b_2 * \text{OM} + b_3 * D + \ldots + b_x * X$, where C is percent clay, OM is percent organic matter, D is bulk density, and X any other basic soil property that can easily be determined. Parameters b_0 through b_x are determined by regression of θ at for example $h = -100$ cm versus relevant soil properties. A disadvantage of using continuous PTFs for predicting specific points of interest is that a large number of different functions are required to describe the complete θ–h–K relationship. This feature strongly hampers the efficient inclusion of hydraulic characteristics in simulation models. Alternatively, parameterization methods are used that predict parameters in a model describing the θ–h–K relationship. This latter approach is more efficient than the point-prediction procedure.

To derive continuous PTFs for soils in the Netherlands, all individually measured hydraulic characteristics were parameterized using the equations of van Genuchten (1980). After parameterization, linear regression was used to investigate the dependency of each model parameter on more easily measured basic soil properties. To comply with a number of physical boundary conditions, transformed parameters rather than the original model parameters are used in the regression analysis. In the case of sandy soils the imposed boundary conditions are: $K_s > 0$, $\alpha > 0$, $n > 1$, and $-2 < l < +2$. In the case of loamy and clayey soils the latter boundary condition is $-10 < l < +10$. As a consequence, parameters are transformed as follows: $K_s^* = \ln(K_s)$, $\alpha^* = \ln(\alpha)$, $n^* = \ln(n - 1)$; for sandy soils $l^* = \ln((l + 2)/(2 - l))$, and for loamy and clayey soils, $l^* = \ln((l + 10)/(10 - l))$. For sandy soils the following basic soil properties are used as regressed variables: percent clay + silt; percent organic matter; bulk density; median sand particle size; and also the qualitative variable topsoil or subsoil. For loamy and clayey soils, the regressed variable percent clay + silt is replaced by percent clay.

Linear, reciprocal, and exponential relationships of these basic soil properties are used in the regression analysis, and possible interactions are also investigated. As a consequence, the resulting regression model or continuous PTF consists of various basic soil properties and their interactions, all of which contribute significantly to the description of the transformed model parameters. This model is selected with the subset selection method of Furnival and Wilson (1974). The resulting continuous PTFs are presented in Table 10.1. After predicting the transformed model parameters with these functions, the hydraulic characteristics are obtained by back-transformation to the original model parameters.

A comparable approach to derive continuous PTFs for different soils has been employed by a number of researchers (e.g., Ghosh, 1980; Cosby et al., 1984; Rawls and Brakensiek, 1985; Saxton et al., 1986; Gregson et al., 1987; Vereecken et al., 1989, 1990). Tietje and Tapkenhinrichs (1993) compared 13 different PTFs with respect to their applicability and accuracy in predicting measured water retention functions. The evaluated PTFs consisted of point-prediction methods and model parameter prediction methods for the prediction of the water retention function. They concluded that the latter method using the empirical equation of van Genuchten (1980) was the most practical and the most accurate approach. It was emphasized that prediction procedures performed poorly if used for soils that fall outside the texture range of soils originally used to derive the PTFs. Therefore, the use of PTFs to predict hydraulic characteristics should be confined to soils with a texture within the range of textures of soils that were originally used to derive the PTFs.

The same set of Dutch soils used to derive continuous PTFs for the prediction of hydraulic characteristics was used also to derive continuous PTFs for the prediction of the bulk density of the soil (actually the reciprocal value or specific volume in $cm^3\ g^{-1}$ was predicted). Use of the same statistical techniques resulted in two continuous PTFs, one for sandy soils and one for loamy and clayey soils, that predict the specific volumes of these soils (Table 10.1).

An attractive feature of continuous PTFs is that the uncertainty involved in using them can be quantified. Wösten and van Genuchten (1988) expressed this uncertainty in terms of 90% confidence intervals (Fig. 10.3). They concluded that, because the confidence intervals are relatively wide, predictions made with their PTFs will show considerable dispersion. Alternatively the PTFs can be used to predict model parameters for the same set of soils from which the PTFs were derived. Differences between "measured" and "predicted" model parameters can be expressed in a variance–covariance matrix. In turn, this matrix allows prediction of average hydraulic characteristics of a soil for which no measured characteristics are available, as well as a quantification of uncertainty of the predicted average characteristics. More details on this approach are provided by Vereecken et al. (1992) and Finke et al. (1996). The latter researchers used the PTFs not only to predict the mean hydraulic characteristics of a soil but also to predict 20 hydraulic characteristics expressing the

TABLE 10.1

Continuous pedotransfer functions for the prediction of hydraulic characteristics and bulk density of soils in the Netherlands

Continuous PTFs for sandy soils:	
$\theta_s = -13.6 - 0.01533{*}CS + 0.0000836{*}CS^2 - 0.0973{*}CS^{-1} + 0.708{*}D^{-1} - 0.00703{*}M50 + 225.3{*}M50^{-1} + 2.614{*}\ln(M50) + 0.0084{*}OM^{-1} + 0.02256{*}\ln(OM) + 0.00718{*}D{*}CS$	$(R^2 = 71\%)$
$K_s{*} = 9.5 - 1.471{*}D^2 - 0.688{*}OM + 0.0369{*}OM^2 - 0.332{*}\ln(CS)$	$(R^2 = 32\%)$
$\alpha{*} = 146.9 - 0.0832{*}OM - 0.395{*}topsoil - 102.1{*}D + 22.61{*}D^2 - 70.6{*}D^{-1} - 1.872{*}CS^{-1} - 0.3931{*}\ln(CS)$	$(R^2 = 53\%)$
$l{*} = 0.797 - 0.591{*}OM + 0.0677{*}OM^2 + 0.573{*}topsoil$	$(R^2 = 42\%)$
$n{*} = 1092 + 0.0957{*}CS + 1.336{*}M50 - 13229{*}M50^{-1} - 0.001203{*}M50^2 - 234.6{*}\ln(M50) - 2.67{*}D^{-1} - 0.115{*}OM^{-1} - 0.4129{*}\ln(OM) - 0.0721{*}D{*}CS$	$(R^2 = 63\%)$
$1/D = -1.984 + 0.01841{*}OM + 0.032{*}topsoil + 0.00003576{*}CS^2 + 67.5{*}M50^{-1} + 0.424{*}\ln(M50)$	$(R^2 = 72\%)$
Continuous PTFs for loamy and clayey soils:	
$\theta_s = 0.8085 - 0.2617{*}D - 0.038{*}topsoil + 0.00001046{*}C^2 + 0.01287{*}\ln(OM) + 0.000789{*}C{*}topsoil$	$(R^2 = 86\%)$
$K_s{*} = -43.1 + 64.8{*}D - 22.21{*}D^2 + 7.02{*}OM - 0.1562{*}OM^2 + 0.985{*}\ln(OM) - 0.01332{*}C{*}OM - 4.71{*}D{*}OM$	$(R^2 = 30\%)$
$\alpha{*} = 11 - 2.298{*}D^2 - 12.41{*}D^{-1} + 0.838{*}OM + 0.343{*}OM^{-1} + 2.03{*}\ln(OM) - 1.263{*}D{*}OM$	$(R^2 = 51\%)$
$l{*} = 0.451 + 2.678{*}D^{-1} - 1.093{*}\ln(C)$	$(R^2 = 44\%)$
$n{*} = -0.34 + 1.224{*}D^{-1} - 0.7952{*}\ln(C) - 0.3201{*}\ln(OM) + 0.0651{*}D{*}OM$	$(R^2 = 74\%)$
$1/D = 0.603 + 0.003975{*}C + 0.00207{*}OM^2 + 0.01781{*}\ln(OM)$	$(R^2 = 77\%)$

θ_s, $K_s{*}$, $\alpha{*}$, $l{*}$ and $n{*}$ are the transformed model parameters in the van Genuchten–Mualem equations; C = percent clay (i.e. percent <2 μm); CS = percent clay + silt (i.e. percent < 50 μm); OM = percent organic matter; D = bulk density; M50 = median sand particle size; topsoil and subsoil are qualitative variables having the value of 1 or 0 and ln = the natural logarithm.

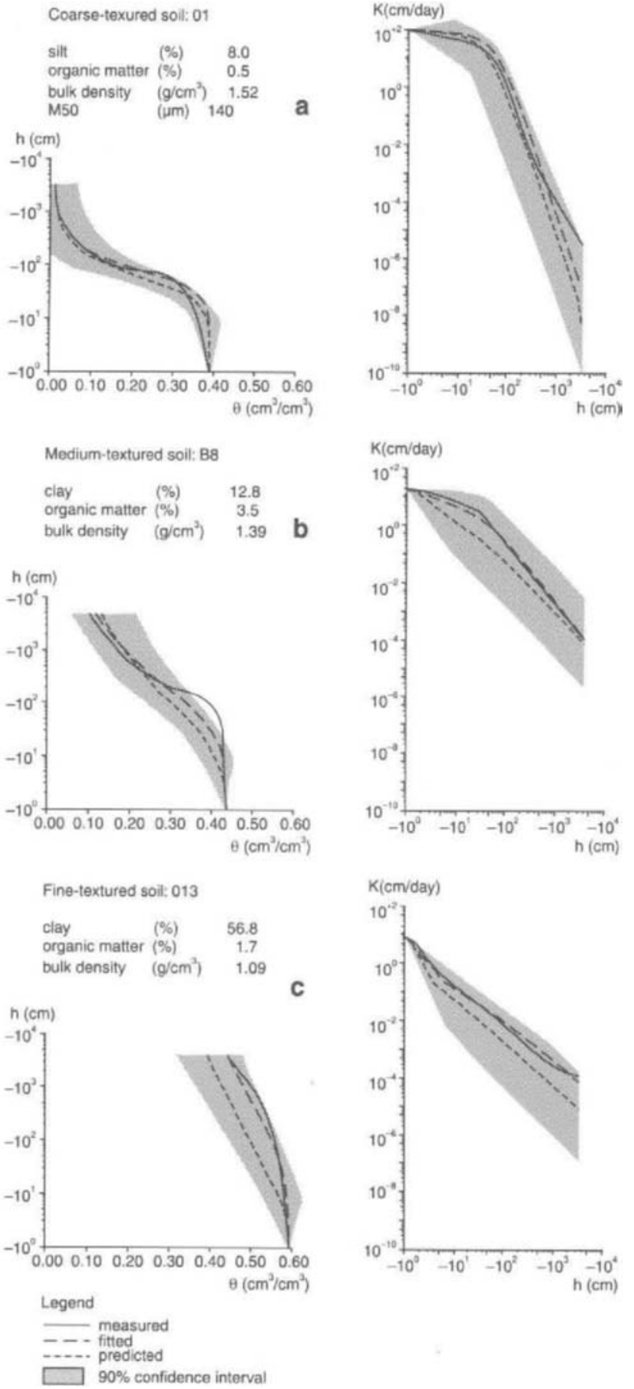

Fig. 10.3. Measured, fitted, and predicted water retention and hydraulic conductivity characteristics for three typical coarse-, medium- and fine-textured soils (after Wösten and van Genuchten, 1988).

uncertainty in the applied PTF. In turn, they used the 20 hydraulic characteristics in a Monte Carlo procedure to calculate a number of different aspects of functional soil behaviour. Their approach showed to what extent variability in calculated aspects of soil behaviour is explained by uncertainty in PTFs.

IV. USE OF THE TWO TYPES OF PEDOTRANSFER FUNCTIONS: A CASE STUDY

Hydraulic characteristics are essential input data for the calculation of water and solute transport in the unsaturated zone. To demonstrate this a case study is presented in which the class- and continuous-PTF approach are used to predict the soil hydraulic input data for simulation of four functional aspects of soil behaviour.

A. Introduction

The effects of using either class or continuous PTFs were investigated by Wösten et al. (1995). They used both approaches to generate soil hydraulic characteristics and then used these characteristics to calculate the following four functional aspects of soil behaviour, the first two of a "physical" nature and the last two of a "chemical" nature:

(i) The number of workable days, which was defined as the number of days between 15 March and 15 May 1980, during which the pressure head, h, at 5 cm below soil surface was ≤ -70 cm;
(ii) The number of days with sufficient aeration, which was defined as the number of days in the same time period with an air-filled porosity at 5 cm depth ≥ 0.1 cm^3 cm^{-3};
(iii) The amount leached after one year of the adsorbing, inert contaminant cadmium (Cd);
(iv) The amount leached after one year of the adsorbing, degrading herbicide Isoproturon.

For the simulated functional aspects, no independently measured values were available. Therefore the two approaches were evaluated only on a relative and not an absolute scale. Simulations of water and solute movement were made with a one-dimensional, finite-difference model that described transient, unsaturated water and solute flow in a heterogeneous soil/root system that may or may not have been under groundwater influence. Calculations were made for a bare soil for a period of 18 months, from 1 October 1979 to 1 April 1981, for a set of 88 soil profiles. This set of profiles was used as a statistically representative set for sandy soils in the north-eastern part of the Netherlands. The soils occupy an area of 115 km^2 and were developed in aeolian sand sediments. Breakthrough of the two solutes was calculated for an imaginary plane at 40 cm below soil surface. It was considered that solutes that pass this plane are lost for plant uptake and eventually migrate to the groundwater.

Soil physical input data for every horizon of the 88 soil profiles were obtained using both the class- and continuous-PTF approach. Recorded soil data for every horizon allowed for the prediction of soil hydraulic characteristics with the two approaches. Required upper and lower boundary conditions for simulation were obtained from field observations.

B. Results and discussion

Based on the calculated functional aspects for each profile, an areal weighted average was calculated for the whole 115 km^2 occupied by the mapping unit. Table 10.2 shows the means and standard errors of the calculated four functional aspects of soil behaviour for the area as a whole. Differences in results using class and continuous PTFs were statistically tested using a t-test on differences between paired samples, taking into account the applied sampling design.

Figure 10.4a shows the number of workable days in the 62-day period from 15 March to 15 May 1980, calculated both with class and continuous PTFs to predict soil hydraulic characteristics. The figure shows that for most of the 88 soil profiles, more than 55 days in the considered period were workable. At the same time, results obtained with the class-PTF approach agree generally well with those obtained with the continuous PTF. In this case, differences between the two approaches were not statistically significant (Table 10.2). The same result was obtained when the threshold value is changed from $h \leq -70$ cm to $h \leq -90$ cm.

Figure 10.4b shows the number of days with good aeration. Using the class-PTF approach, practically all 88 profiles had sufficient aeration for more than 55 out of the 62 days. However, when continuous PTFs were used, the number of days with adequate aeration was highly variable, ranging from 0 to 62 days for the 88 profiles. In this case differences between the two approaches were significant (Table 10.2). It appears that when the number of days with adequate aeration is considered, class

TABLE 10.2

Areal weighted values for four functional aspects of soil behaviour calculated for 88 soil profiles using both class and continuous pedotransfer functions (PTFs) to predict soil hydraulic characteristics (after Wösten et al., 1995)

Functional aspect	Continuous PTF		Class PTF		Test on differences			
	m	s.e.	m	s.e.	m	s.e.	Lower and upper 95% confidence limit	
Workable days	57.9	0.8	57.8	0.8	−0.2	0.35	−0.9	0.54
Days with good aeration	53.8	1.8	60.8	0.5	7.0	1.44	4.1	9.9
Log leaching mass cadmium	−8.50	0.12	−8.48	0.12	0.02	0.004	0.01	0.03
Log leaching mass Isoproturon	−5.30	0.24	−5.28	0.24	0.02	0.008	0.00	0.03

m = mean value, s.e. = standard error.

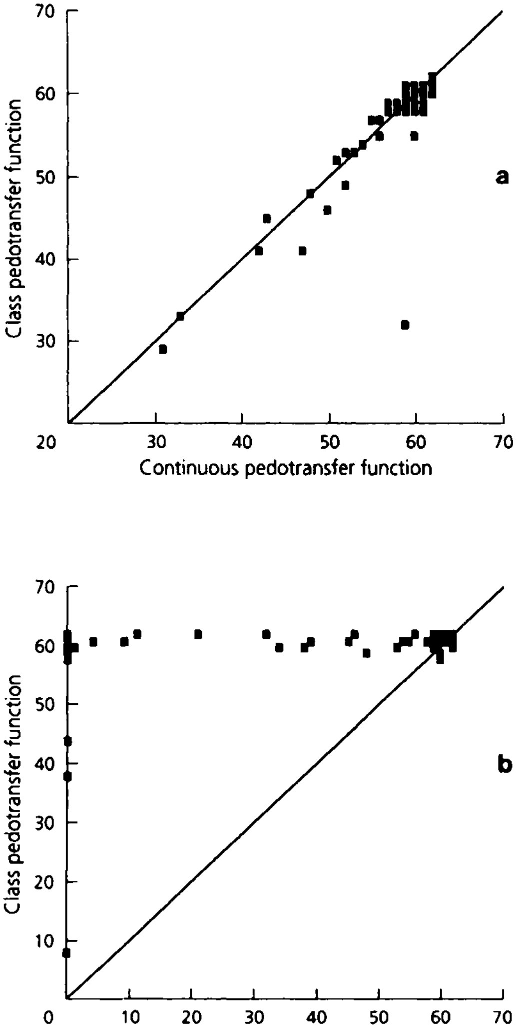

Fig. 10.4. Relationship between the number of workable days (a) and number of days with good aeration (b) calculated using class and continuous pedotransfer functions to predict soil hydraulic characteristics (after Wösten et al., 1995).

PTFs fail to generate hydraulic characteristics sufficiently different to affect this functional aspect. The same holds true when the threshold value for the air-filled porosity is changed from ≥ 0.1 cm^3 cm^{-3} to ≥ 0.15 cm^3 cm^{-3}. It was concluded that different results between the class- or continuous-PTF approaches depend on the specific physical functional aspect of soil behaviour under consideration. Thus, in order to calculate these different functional aspects, emphasis should be placed on different parts of the soil hydraulic characteristics (i.e., dry versus wet soil).

Figures 10.5a and 10.5b show the results for the chemical functional aspects, the leaching mass of Cd and of Isoproturon expressed in kg ha^{-1}. Since the values are log-normally distributed, the log values are presented. In both cases results obtained with class PTFs seem to agree well with those obtained with continuous PTFs. However, differences between the two approaches were statistically significant (Table 10.2). The amount of leached Cd and Isoproturon was always slightly higher when continuous PTFs were used to estimate the soil hydraulic characteristics. In practice, these differences were so small that they are irrelevant. In the case of solute

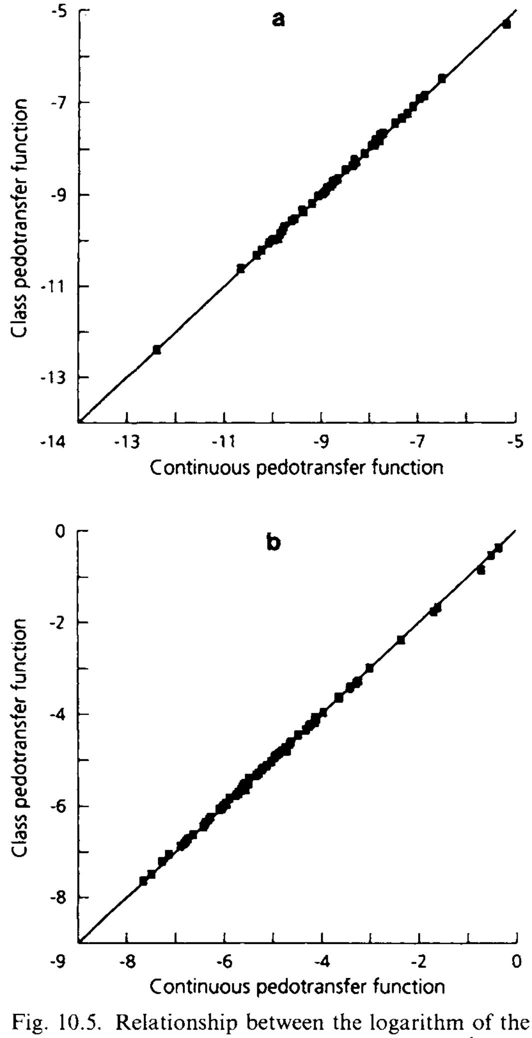

Fig. 10.5. Relationship between the logarithm of the leaching mass (kg ha^{-1}) of cadmium (a) and the logarithm of the leaching mass (kg ha^{-1}) of Isoproturon (b) calculated using class and continuous pedotransfer functions to predict soil hydraulic characteristics (after Wösten et al., 1995).

transport, the adsorption and decay processes apparently diminish the differences that occur when only soil physical processes are active.

C. Conclusions

The major conclusion from this study is that when functional aspects of soil behaviour are calculated, results can be different depending on whether class or continuous PTFs are used to generate the hydraulic input data. Therefore, the choice between the class- or continuous-PTF approaches is ambiguous and depends on the functional aspect of interest. When adsorption and decay processes are active, they diminish the differences that can be attributed to the use of the class or continuous PTFs.

As a result, it is useful to use existing data bases of measured hydraulic characteristics for the establishment of both class and continuous PTFs. In cases where both approaches yield the same result, easy-to-use and less expensive class PTFs are preferred over the more laborious continuous PTFs. Independently measured functional aspects of soil behaviour are required to assess the accuracy of the two pedotransfer approaches.

V. USE OF PEDOTRANSFER FUNCTIONS TO PREDICT OTHER THAN SOIL HYDRAULIC CHARACTERISTICS

So far this discussion on PTFs has focused on the transfer of easy-to-obtain soil survey information into soil hydraulic characteristics. However, the same soil survey data can be used to make predictions for soil chemical and soil fertility properties. The following examples are presented to show how PTFs were developed for other soil properties.

Bakker et al. (1987) presented PTFs for the prediction of the oxygen-diffusion coefficient in soils (D_s), expressed in m^2 s^{-1} at 20°C, from the volume fraction of gas ($_gV$) in the soil. They differentiated their PTFs for Dutch soils according to soil structure and soil texture as follows:

$D_s = 0.3\ 10^{-4}\ V_g^{3.0}$ for single-grain structured sandy soils; (4)

$D_s = 1.5\ 10^{-4}\ V_g^{4.0}$ for single-grain structured loamy sands and sandy loams; (5)

$D_s = 0.5\ 10^{-4}\ V_g^{3.0}$ for weakly and moderately aggregated topsoils of silt loams and humic sands as well as for subsoils of silt loams (6)

$D_s = 0.4\ 10^{-4}\ V_g^{2.5}$ for clearly aggregated silt loams and clays; (7)

$D_s = 0.06\ 10^{-4}\ V_g^{1.5}$ for compacted silt loams and clays. (8)

These PTFs are not valid in soils that show a tendency to shrink and swell.

Breeuwsma et al. (1986) predicted the cation exchange capacity (CEC, expressed in mmol g^{-1} soil) of different soil horizons in the Netherlands as follows:

$$CEC = 2.5\ OM + 0.5\ C \qquad \text{for B horizons of Haplaquods;} \tag{9}$$

$$CEC = 1.5\ OM + 0.5\ C \qquad \text{for other soil horizons and/or soil units} \tag{10}$$

where OM and C are the organic matter and clay content in g g^{-1} soil. Based on these PTFs applicable for soil horizons, the CEC expressed in mmol ha^{-1} for the unsaturated zone was predicted using the following PTF:

$$CEC_{\text{unsat.zone}} = \sum_{i=1}^{n} 10^{-1} D_i d_i (a_{OM} OM_i + 0.5\ C_i) \tag{11}$$

where D_i is the thickness of soil horizon i in cm, d_i is the bulk density of soil horizon i in g cm^{-3}, a_{OM} is 2.5 for B horizons and 1.5 for all other soil horizons, and n is the number of soil horizons in the unsaturated zone. In this way soil data were used to predict the required adsorption data for studies of solute movement in soils.

In a similar way Breeuwsma et al. (1986) predicted the phosphate sorption capacity of Dutch soils (PSC), expressed in mmol kg^{-1}, using the following PTF:

$$PSC = 0.4(Al_{ox} + Fe_{ox}) \tag{12}$$

where Al_{ox} and Fe_{ox}, expressed in mmol kg^{-1}, are oxalate-extractable aluminum (Al) and iron (Fe). For soil horizons, the PTF for prediction of the PSC, expressed in kg P_2O_5 ha^{-1}, for the unsaturated zone was as follows:

$$PSC_{\text{unsat.zone}} = \sum_{i=1}^{n} 2.84 D_i d_i (Al_{ox} + Fe_{ox})_i \tag{13}$$

This latter approach was used in the Netherlands to predict the PSC of soils receiving high rates of liquid manure.

Bronswijk et al. (1995) predicted the yearly pyrite oxidation rate (POR), expressed in kg m^{-2} yr^{-1}, in Indonesian acid sulphate soils as a function of the average groundwater level, expressed in cm below soil surface, as follows:

$$POR = 0.0643 \left(\sum_{n=1}^{n=365} \left(h(n) - Z_{FeS2} \right) \right) \Big/ 365 \tag{14}$$

where $h(n)$ is the groundwater level at day n and Z_{FeS2} is the starting depth of the pyritic layer. For days that the groundwater table was above the pyritic layer, $h(n) - Z_{FeS2} = 0$.

In soil fertility studies, Janssen et al. (1990) established PTFs to predict the potential supply of soil nitrogen (SN), phosphorus (SP), and potassium (SK) for maize (*Zea mays* L.). They defined the potential supplies as the maximum quantity of these nutrients that can be taken up by maize if no other nutrients or other factors are limiting. These PTFs have the following forms:

$$\mathrm{SN} = 17 \times (\mathrm{pH} - 3) \times \mathrm{N_{org}}, \text{ or } 1.7 \times (\mathrm{pH} - 3) \times \mathrm{C_{org}} \tag{15}$$

$$\mathrm{SP} = 0.014 \times \left(1 - 0.5 \times (\mathrm{pH} - 6)^2\right) \times \text{total P} + 0.5 \times \text{P-Olsen},$$
$$\text{or, } 0.35 \times \left(1 - 0.5 \times (\mathrm{pH} - 6)^2\right) \times \mathrm{C_{org}} + 0.5 \times \text{P-Olsen} \tag{16}$$

$$\mathrm{SK} = 250 \times (3.4 - 0.4 \times \mathrm{pH}) \times K_{\mathrm{exch}}/(2 + 0.9 \times \mathrm{C_{org}}) \tag{17}$$

The independent regressed variables in these PTFs are the soil chemical properties of the 0–20 cm soil horizon. The PTFs are used as a part of a system for the quantitative evaluation of the native fertility of tropical soils.

Many other PTFs have been developed besides those discussed above. Larson and Pierce (1991) presented a limited list of PTFs applicable in the field of soil chemistry, soil physics, soil hydraulics, and prediction of productivity.

VI. MULTICOLLINEARITY

Continuous PTFs are regression models that use various basic soil properties and their interactions as regressed variables. Complications might occur in deciding which basic soil properties are to be used as regressed variables. This is best illustrated by an example where the PTF has two regressed variables: X_1 and X_2. In this case three situations are possible (Oude Voshaar, 1994):

(i) The part of the sum of squares accounted for by X_1 and X_2 in a combined regression model equals the sum of the sum of squares accounted for by X_1 and X_2 in separate regression models. In this case the regression coefficients in the combined model equal those in the separate models;
(ii) The part of the sum of squares accounted for by X_1 and X_2 in a combined regression model approximately equals the sum of squares accounted for by X_1 and X_2 in the separate regression models;
(iii) The part of the sum of squares accounted for by X_1 and X_2 in a combined regression model is much larger then the sum of the sum of squares accounted for by X_1 and X_2 in the separate models.

In situation (i) the regressed variables X_1 and X_2 are said to be orthogonal. In situation (ii) one regressed variable can be replaced by the other. An example of the latter case is pH determined in water and pH determined in KCl. In this situation, when the regressed variables are strongly correlated, the problem of multicollinearity is said to exist (Montgomery and Peck, 1982). In situation (iii) the regressed variables X_1 and X_2 supplement each other.

In the balanced sampling strategy of situation (i), variables such as the parameters in the van Genuchten equations are predicted correctly and an explanation can be given as to which of the regressed variables has a strong impact on prediction of the model parameter. In this sense the causality is maintained and the interpretation of the effects is unique. However all these features are lost in (ii) and (iii).

Inspection of the correlation matrix of regressed variables is a quick and easy way to determine whether some of the variables are strongly linear dependent. In the latter case it is preferred to replace one variable by the other. Multicollinearity often occurs in unbalanced sample strategies as applied in observational research in which data are taken as they come. However, many interpretation problems can be avoided by aiming at balanced sampling of regressed variables.

VII. CONCLUSIONS

Pedotransfer functions are a powerful tool in estimating physical, chemical, and fertility properties of soils. Because PTFs predict difficult-to-obtain properties from already available basic soil properties, they have the clear advantage that they are relatively inexpensive and easy to derive and use. Accuracy of PTF predictions is sufficient for many applications at regional and national scales since on these scales the effects of temporal and spatial variability most likely will have a dominant impact on the modelling results. Use of PTFs might not be appropriate for application at a specific location, in which case direct measurement is the only option. PTFs should not be used to make predictions for soils that are outside the range of soils used to derive the PTFs originally. In other words, PTFs may be used safely for interpolation but not for extrapolation. PTFs are only as good as the original measured data from which they were derived. This suggests that PTFs should be periodically updated as more measured data become available. To develop useful PTFs it is important that regressed variables in regression models are not strongly linearly related thus preventing multicollinearity.

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Chapter 11

STATISTICAL APPROACHES TO THE ANALYSIS OF SOIL QUALITY DATA

O. WENDROTH, W.D. REYNOLDS, S.R. VIEIRA, K. REICHARDT and S. WIRTH

I. INTRODUCTION

Adequate management of the environment and agricultural resources requires assessment of soil quality. Inasmuch as we want to protect our global resources when managing agricultural systems, we have to achieve efficient use of inputs, such as fertilizers and pesticides, for crop production, as well as avoid pollution risks arising from over-application of agrochemicals. Therefore, spatial and temporal patterns of soil properties and soil quality attributes and indicators have to be known. Moreover, to integrate and regionalize information from specific points to larger scales, knowledge of the nature of spatial and temporal patterns of land surfaces is essential.

In this chapter, we discuss techniques that allow us to analyse field observations statistically. The concept behind these techniques differs from the classical way of conducting agronomic experiments. With new approaches, we no longer have to impose treatments that generally disregard a thorough understanding of the entire agricultural system (Peterson et al., 1993; Nielsen et al., 1994a). Instead, we can derive relevant information directly from on-site observations using tools that manipulate these observations to assess physical, chemical, and biological properties of entire fields (Nielsen et al., 1994b). These monitoring and analyzing methods help focus our attention on the underlying processes that account for the spatial and temporal variability patterns of soils and crops, rather than looking for a significant response to a set of imposed treatments that may not be practicable or even be related to optimal management practices for a particular field (Nielsen and Alemi, 1989).

When farmers manage their fields, they intuitively pay attention to local soil variability within their fields—information that has often been suppressed in agronomic studies. Now we have the opportunity to expand the intuitive thoughts of the farmer using analytical tools to provide better management alternatives designed specifically for each of his particular fields.

The objectives of this chapter are to illustrate basic principles, aspects, and requirements for spatial statistical analyses and to present some applications of selected geostatistical techniques (Isaaks and Srivastava, 1989) and time series analysis (Shumway, 1988). This contribution should allow answers to the questions: For a given area such as a farmer's field or watershed, what are the patterns of soil properties, crop attributes, and yields that display spatial variability? How can these patterns be identified and understood? How can this identification and understanding be used to optimize profitability and agricultural and environmental sustainability?

II. APPROACHES

A. *Autocovariance*

Usually when a variable is sampled in the field, the mean and the variance are determined to reflect the sampled population, assuming that sampling occurred randomly and representatively (i.e., observations are independent of each other and, in general, are normally distributed).

The set of microbial biomass data sampled along a catena in a landscape ecology study shown in Figure 11.1 has a mean of 626.8 μg C g^{-1} and a variance of 6811.1 (μg C $g^{-1})^2$. Fifty samples were taken at 1.8-m intervals along a 90-m long transect. The following analysis shows additional information that a spatial analysis of a univariate data set can provide when sampling coordinates are considered, rather than ignoring them as is commonly done in most agronomic experiments. The autocovariance-lag distance function or simply the autocovariance function $C(h)$ is defined as

$$C(h) = \frac{1}{N}\sum_{i=1}^{N-h}(x_{i+h} - \bar{x})(x_i - \bar{x}) \tag{1}$$

where a set of N observations x_i at location i with mean $\bar{x}$ is considered (Salas et al., 1988). The distance between pairs of observations is h, the so-called lag. When C(h) is normalized (i.e., it is divided by the sample variance s^2) it is called the autocorrelation function $r(h)$, which is bounded between +1 and −1, and is determined by:

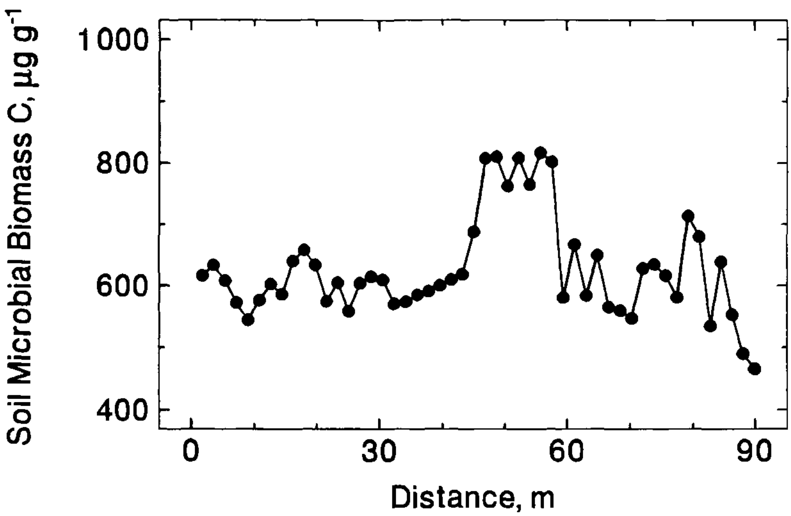

Fig. 11.1. Soil microbial biomass C along a transect sampled across a moraine catena in NE Germany.

$$r(h) = \frac{C(h)}{s^2} \tag{2}$$

where

$$s^2 = \frac{\sum_{i=1}^{N}(x_i - \bar{x})^2}{N - 1} \; . \tag{3}$$

In Figure 11.2a, the autocorrelation function for the biomass data is shown. We can obtain $r(h)$ by plotting the observations x_i against the observations x_{i+h}, and can then calculate the respective correlation coefficient $r(h)$ for this scatter diagram. With increasing lag distance, the number of pairs x_i versus x_{i+h} decreases, and therefore the reliability of $r(h)$ becomes small for widely separated observations (large h-values) unless N is very large.

Another tool reflecting the autocovariance versus lag relation is the semivariogram or simply variogram $\gamma(h)$, calculated according to the following:

$$\gamma(h) = \frac{1}{2N(h)}\sum_{i=1}^{N(h)}(x_i - x_{i+h})^2 \tag{4}$$

where $N(h)$ is the total number of sample pairs for the lag interval h. For the chosen sampling distance (h), half of the average squared difference between all pairs of observations separated by that distance is calculated. Unlike the autocorrelation function, the semivariogram is not based on the total sample variance but on variation between pairs of observations. Hence, it is not bound so strictly to stationarity assumptions as is the autocorrelation function. Stationarity means that

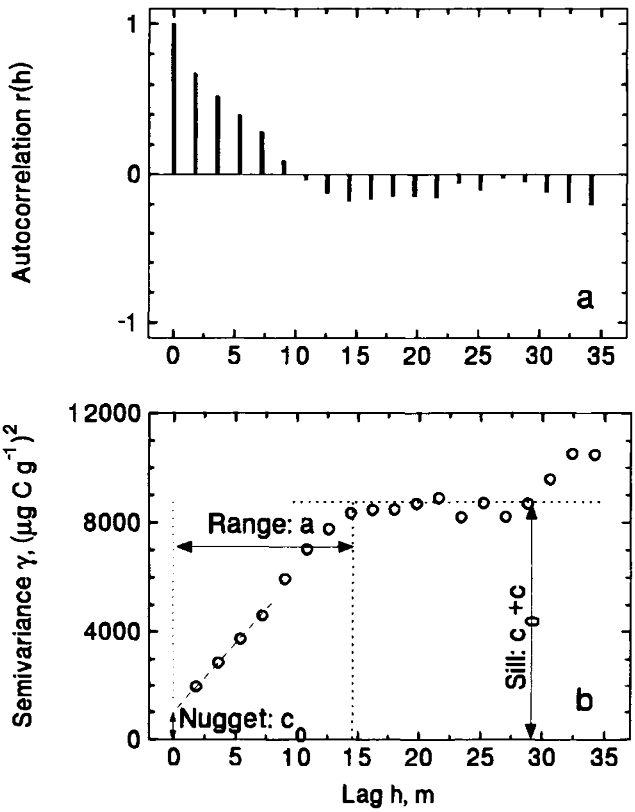

Fig. 11.2. Autocorrelation function (a), and semivariogram (b) for microbial biomass C data.

the mean and the variance of the data do not change appreciably within the sampled region, which implies as a consequence that there are no overall trends or slopes in the data values with position. When stationarity exists the semivariogram (Fig. 11.2b) is a mirror image of the autocorrelation function. The semivariogram can be used for spatial interpolation purposes such as kriging (as shown in the example below for land evaluation with respect to atrazine leaching).

Between 1.8 and 5.4 m (or between 1 and 3 lags) in Figure 11.1, the autocorrelation of biomass data decreases, and the semivariance increases steeply (Fig. 11.2a,b). The zone of increasing semivariance is called the *range*. The range, which reflects the structured variability of observations, is 14.4 m or 8 lags in our example (Fig. 11.2b). Hence, we can say that up to a distance of roughly 14 m, microbial biomass observations are correlated with each other. When the semivariance does not change significantly with increasing lag distance, the plateau reached is called the *sill*, reflecting the magnitude of random variation, which in our example is around 8500 (μg C $g^{-1})^2$.

Although by definition $\gamma(h)$ equals 0 at $h = 0$ (i.e., the variability of the measured parameter is zero at zero distance from the location of measurement), the *nugget* is the effective value of $\gamma(h)$ at $h = 0$ extrapolated from values of $\gamma(h)$ for $h > 0$. The nugget reflects that fraction of the variance at the shortest sampling distance, that is attributed to measurement uncertainty (human error, measurement error, repeatability, etc.) and nested structures having ranges smaller than the sampling interval (Olea, 1991). In some cases, the magnitude of the nugget can be reduced by sampling at shorter h intervals. In the case of our example of biomass determination (Fig. 11.1), errors may arise from sample augering and preparation, calibration of the analyser (CO_2-detector), and measurement noise. The latter may be reduced by repeated measurement with the same sample if possible. In view of the small nugget variance in our example (approximately 1000 $(\mu g\ C\ g^{-1})^2$ and only 12% of the sample variance, Fig. 11.2b), the method for determining biomass is considered sufficiently reliable to allow satisfactory determination of its spatial structure of variation.

Up to a distance of 14.4 m, observations are spatially correlated (Fig. 11.2b) (i.e., having sampled at a location, the variance band is known for the expectation of a biomass value at a certain location by knowing neighboring values and their separation distances to that location). One may conclude from the biomass semivariogram that a structured variability could have been identified even if the separation distance between nearest samples were increased. On the other hand, if samples had been taken at distances greater than 15 m across this 90-m transect, they would have appeared to vary randomly in space. The effect would have been the same as if their coordinates had been neglected, namely no information would have been gained on representativity of the sampling for biomass or the spatial continuum of microbial biomass. In that case, no spatial interpolation would be possible. Moreover, if the investigator had sampled randomly and had by chance received only one instead of seven samples from the zone between 45 and 58 m (Fig. 11.1), where an underlying but unidentified process apparently caused high microbial biomass values, this sample value would probably have been interpreted as an outlier. But having gained seven samples close to each other yielding higher biomass values, and knowing their sampling locations, a degree of certainty is given to the investigator that a sampling or measuring error had not occurred, and that a process not yet determined caused higher values in that region of the transect. Next, a study could follow, investigating: 1) how the spatial pattern of this parameter looks at a different sampling time; and 2) whether the spatial pattern of microbial biomass is linked to other soil and agronomic properties.

B. Crosscovariance

Similar to the autocovariance, the spatial relation between different variables can be determined via the crosscovariance, namely the crosscorrelation function $r_{xy}(h)$ (CCF) and the cross-variogram $\Gamma(h)$. The crosscovariance as a function of lag distance describes the degree of linkage between two variables x and y, where one variable, the tail variable, lags behind the head variable by the lag distance h (see Davis, 1986; Shumway, 1988). The CCF is unsymmetric, whereas the crossvariogram (or covariogram) is symmetric. Because the $\Gamma(h)$ function is symmetric, it produces the same result regardless of whether the x variable is heading or tailing.

In the following example, the crosscorrelation function is calculated for almond (*Prunus amygdalus* Batsch) yields, measured for each of 62 trees within two neighboring parallel transects in an almond orchard north of Sacramento, Calif., U.S.A. The two distributions of almond yield across each transect are similar (Fig. 11.3). The relation between the yields at the same position within the transect is reflected by the classical correlation coefficient, $r = 0.41$, which is also the result for the CCF at lag $h = 0$ (Fig. 11.4). Although the relation between the two variables does not seem to be very tight, the crosscorrelation function (Fig. 11.4) indicates that the observations are spatially related to each other over a distance of about 40 m. In this example the CCF becomes insignificant when $-0.2 < r_{xy}(h) < 0.2$. Hence, although classical correlation indicates a low relation between the variables at the same position, spatial coincidence of both processes is identified when the local range of spatial correlation is examined. Within a certain local range, one variable is certainly related to the other, and this relation can be used for estimating a variable at an unsampled location by means of the spatial crosscovariance structure. In classical regression analysis, the estimation accuracy of a dependent variable depends on the uncertainty of the regression coefficients. Spatial regression techniques, on the other hand, account for correlation structures in the neighborhood of location i, namely within $i \pm h$. Consequently, the estimation uncertainty of a dependent variable can often be reduced substantially by using spatial regression techniques as opposed to classical regression techniques.

Coregionalization procedures or models for spatial realizations of random functions, such as kriging and cokriging, apply this concept. The governing kriging and cokriging equations can be found in Alemi et al. (1988) and Deutsch and Journel (1992). Kriging and cokriging are techniques that are used to estimate unsampled

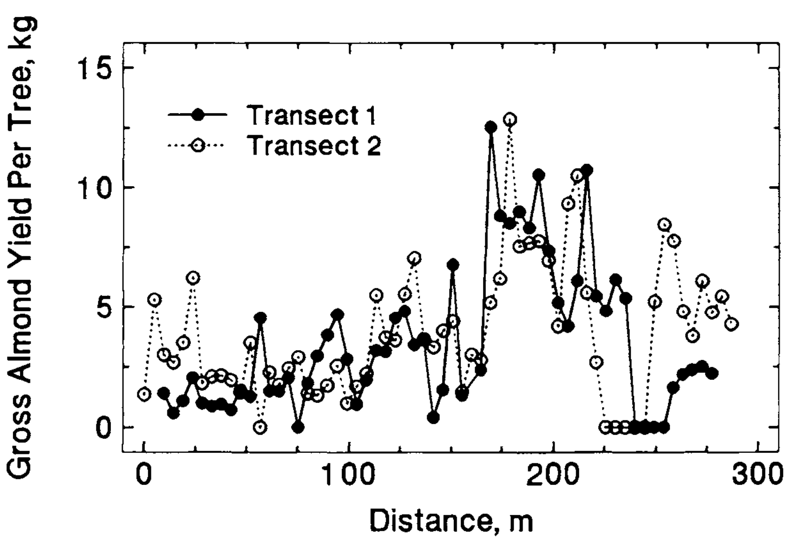

Fig. 11.3. Almond yields across two rows of almond trees in an orchard north of Sacramento, Calif., U.S.A.

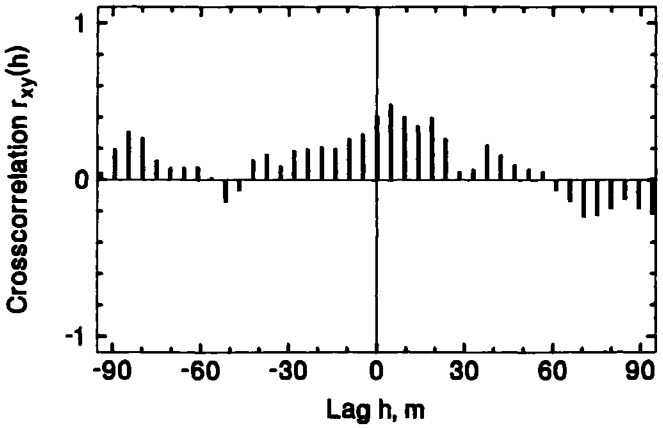

Fig. 11.4. Crosscorrelation function for almond trees in an orchard north of Sacramento, Calif., U.S.A.

data points from observed data. The estimation for an unsampled location is based on known values in the local neighborhood of the unsampled location. The amount that the known values contribute to the estimated value depends on the number of observed points in the vicinity of the location of interest. The weight of the contribution of the known values decreases with increasing distance away from the location of interest. Hence, the estimation is based on a linear combination of the neighborhood values. This estimation differs from that of classical regression analysis in which spatial relations between all locations are ignored, and only one equation reflects causal relations across the entire sampling domain. For kriging and cokriging, significant information obtained from the structure of variation (i.e., the variogram and crossvariogram, respectively) is incorporated as a measure of reliability of the estimation, namely the kriging and cokriging estimation variance (Alemi et al., 1988).

Cokriging can be applied for spatial interpolation, especially in situations in which sampling resources are limited but one needs to gain information about unsampled locations. This technique was recently examined as a multivariate geostatistical tool for yield response and N-pollution by Goovaerts and Chiang (1993). As an example of cokriging, we assume a scenario for the almond yield data across the two parallel transects where yield values are known for only 16 locations in transect 1 but for all locations in transect 2 (Fig. 11.5). In the cokriging procedure, the semivariogram of the variables of interest (Fig. 11.6a,b) and the crossvariogram (Fig. 11.6c) are the underlying information for the estimation variance in the interpolation procedure. The spherical model was chosen to fit the variogram data, as follows:

$$\begin{aligned} \gamma(h) &= c_0 + c\left[1.5\frac{h}{a} - 0.5\left(\frac{h}{a}\right)^3\right], && \text{if } h \leq a \\ &= c_0 + c && \text{, if } h \geq a\ . \end{aligned} \tag{5}$$

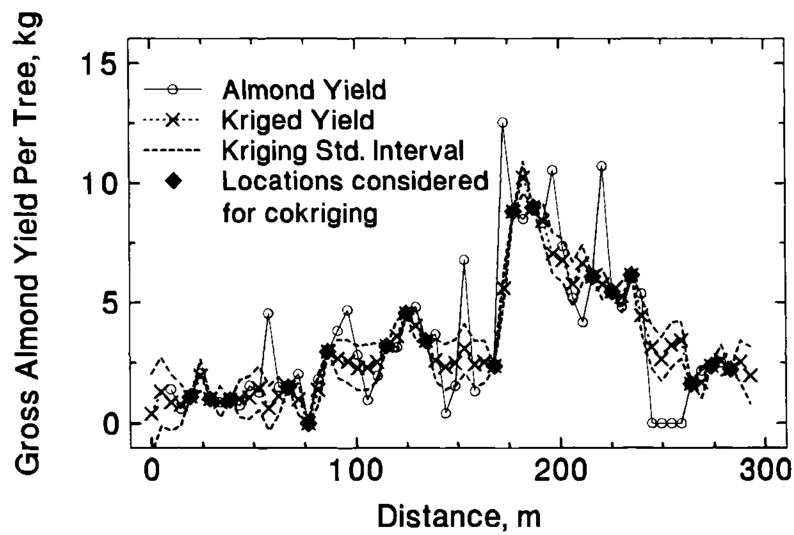

Fig. 11.5. Hypothetical scenario for cokriging of "unknown" almond yield values via spatial covariance.

In equation 5, c_0, c, a denote the nugget, sill, and the range (see Fig. 11.2), respectively. Because values of $\gamma(h)$ and $\Gamma(h)$ are most important at short distances, the observations in transect 1 were not spaced equally but in a nested structure (i.e., nests with short sampling distances were distributed over the entire length of the transect). The use of nested sampling allows a relatively small number of observations to adequately determine the semivariance at small sampling distances, and thereby reduce the nugget.

Like any other interpolator, cokriging smooths the spatial process of data and tends to fail especially at large fluctuations between neighbors. The standard estimation error intervals shown in Figure 11.5 become wider with increasing distance from the formerly observed point and decrease again when approaching the next observed point. Note that with classical regression the fiducial limits of estimation would be wider than for kriging, and also constant, regardless of the proximity of an observed point.

In this one-dimensional example (Fig. 11.5), the power of cokriging cannot be fully described. In two- or three-dimensional sampling designs, cokriging can be used for mapping purposes and can help to estimate patterns of variables based on the crosscovariance with "cheaper" variables. It can be used for coregionalization of different variables observed at the same location (or in a parallel array of locations, such as in our example of the two neighboring transects) and for prediction of spatial patterns at different times (e.g., relating crop yield patterns to those of soil water content; Bouten et al., 1992). Cokriging even allows anisotropic variation structure to be accounted for (i.e., when variograms have different slopes or shapes for different directions in space). At this point the more interested reader is referred to

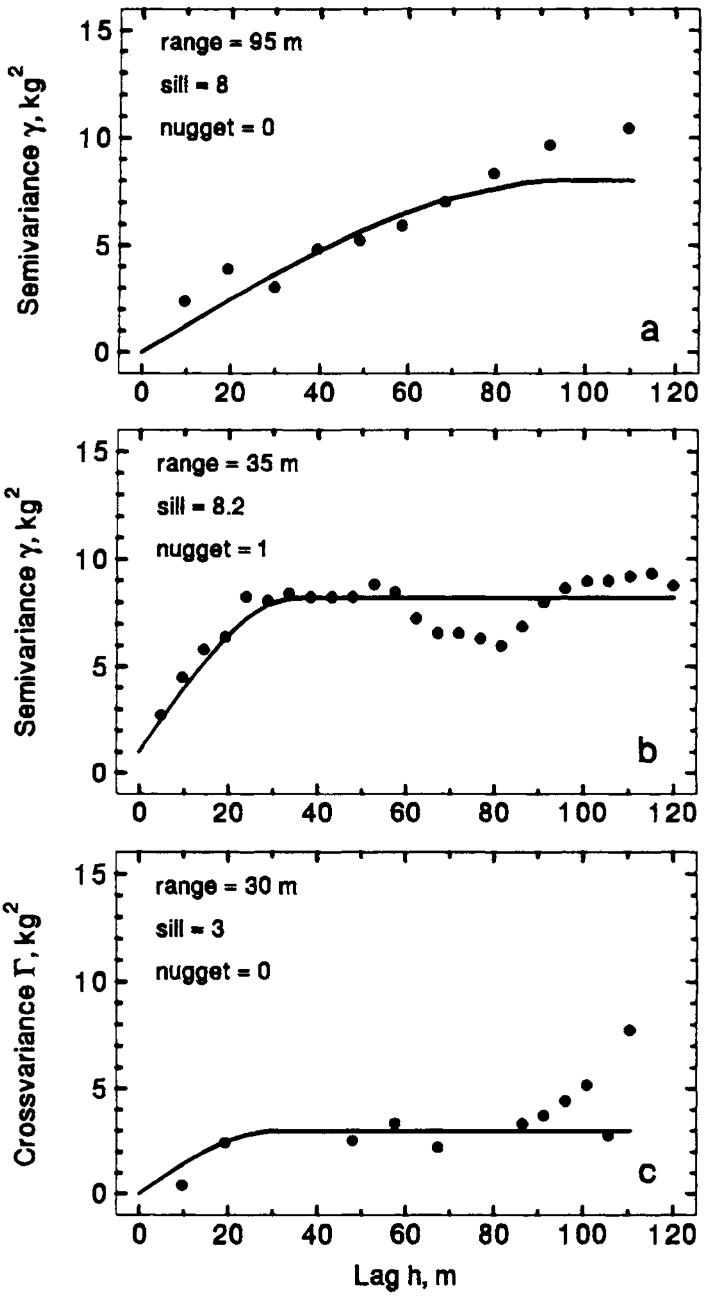

Fig. 11.6. Semivariograms for almond yields in row 1 (a), row 2 (b), and the crossvariogram (c). Parameters for spherical variogram model are given.

the literature (e.g., Alemi et al., 1988; Deutsch and Journel, 1992; Zhang et al., 1995; Halvorson et al., 1995).

Nevertheless, kriging and cokriging are dependent on the validity of the variograms and crossvariograms upon which they rely. Additionally, stationarity assumptions have to be met for kriging and cokriging applications, whereas other interpolation tools, such those presented in the following section, are not necessarily limited to stationarity conditions (Shumway, 1985).

C. *State-space analysis*

The state-space analysis (Shumway, 1988) demonstrated here is a special kind of autoregressive model. State-space analysis can be used, like kriging and cokriging, for spatial interpolation but the philosophy behind this tool is different from that of kriging. In state-space analysis, a system's state (i.e., the state of a variable or of a set of variables) at location i, is considered with respect to the system's state at location $i - h$, where $h = 1,2,3,\ldots, n - 1$. These kinds of autoregressive tools are used for various kinds of forecasting based on the process of a series through the past to identify the coefficients linking system's states (the state coefficients) through space or time. Economical time series, remote controlled missiles, soil temperature and water content series (Morkoc et al., 1985a), crop yield and soil nitrogen status (Wendroth et al., 1992), and lake water storage (Assouline, 1993) are a few examples of data modeled with state-space approaches.

The basic equation, the so-called state equation, is as follows:

$$Z_i = \Phi Z_{i-1} + \omega_i \tag{6}$$

where Z_i is the state vector (i.e., a set of p variables at location i), Φ is a $p * p$ matrix of state coefficients indicating the measure of spatial regression, and ω_i is the uncorrelated zero mean model error. So far, this is the usual structure of common autoregressive models, where coefficients in the Φ matrix could be calculated via multiple regression. Here, Z_i is equivalent to the dependent and Z_{i-1} to the independent variable, respectively. Unlike common autoregressive models, however, the "true" state of the variable or of the state vector in state-space models is considered embedded in the following observation equation:

$$Y_i = M_i Z_i + ny_i \tag{7}$$

where the observed vector Y_i is related to the true state vector Z_i via an observation matrix M_i and an uncorrelated mean zero observation error ny_i. In other words, what is measured does not have to be fully taken to be true, but can be considered as an "indirect measure" reflecting the "true" state of the variable plus noise (unidentified error). This error is associated with measurement uncertainty arising from reproducibility and validity of the calibration underlying the "indirect" observation. Note that almost every observation has to be considered as an indirect measure.

Moreover, unlike common autoregressive modeling, the state coefficient and covariance matrices are optimized via Kalman filtering (Kalman, 1960) within an iterative algorithm. Unlike multiple regression, the Kalman filter accounts for measurement and model errors by not taking the measurements to be absolutely true but allowing for the variance of the state. In the state-space coefficient estimation, 1) the value at step i is predicted based on the state at $i-1$ and a given set of coefficients, 2) the prediction is compared to the measurement, and 3) the prediction is updated as far as the deviation between measurement and prediction requires, while accounting for both model and measurement errors. Steps 1 to 3 are repeated, while coefficients are optimized iteratively, until a convergence criterion is met. For further details, see Shumway (1988), Katul et al. (1993) and Nielsen et al. (1994a).

In the following example, observations of a field study from the International Atomic Energy Agency (IAEA) experimental field in Seibersdorf, Austria, described in Reichardt et al. (1987) were analyzed using a state-space approach (see also Wendroth et al., 1992). In two neighboring transects, a field experiment was established in order to estimate spatial variation of symbiotic nitrogen fixation of a legume crop, alfalfa (*Medicago sativa* L.). Soil and crops were sampled every 1.8 m across a 96-m long transect. The heterogeneous soil contained a considerable volume fraction of small stones that varied across the site (Fig. 11.7a). Knowing that soil nitrogen content affects both crop production and the rate of symbiotic nitrogen fixation, the soil nitrogen has to be considered on a volume basis as effective nitrogen *N*eff (Fig. 11.7b). Therefore, the volume of stones per unit soil volume was accounted for in the calculations. Based on the ^{15}N-isotope dilution method, nitrogen fixed under the alfalfa crop was determined (Fig. 11.7c). The crop yields of ryegrass (*Lolium* sp.) and alfalfa are shown in Figure 11.8.

In a scenario where we assume to know every *N*eff value, but only a cyclic sequence of three ryegrass yield observations followed by three unknown values the coincidence of *N*eff and ryegrass yield processes was determined. The result of the state-space estimation with the respective state equation is presented in Figure 11.9a. The underlying system of equations is as follows:

$$\begin{pmatrix} RY_i \\ N\mathrm{eff}_i \end{pmatrix} = \begin{pmatrix} \phi_{11} & \phi_{12} \\ \phi_{21} & \phi_{22} \end{pmatrix} \begin{pmatrix} RY_{i-1} \\ N\mathrm{eff}_{i-1} \end{pmatrix} + \begin{pmatrix} \omega_{RYi} \\ \omega_{N\mathrm{eff}_i} \end{pmatrix} \tag{8}$$

i.e., the ryegrass yield at location i is determined as a function of ryegrass yield and *N*eff, both at location $i-1$, plus a model error ω. The estimated state coefficient matrix incorporates the spatial regression between neighboring locations as well as the effect due to measurement noise. For the alfalfa crop, both *N*eff and the nitrogen derived from the atmosphere (*N*dfa) caused variation of crop yield. In this case the equations take the form:

$$\begin{pmatrix} AY_i \\ N\mathrm{eff}_i \\ N\mathrm{dfa}_i \end{pmatrix} = \begin{pmatrix} \phi_{11} & \phi_{12} & \phi_{13} \\ \phi_{21} & \phi_{22} & \phi_{23} \\ \phi_{31} & \phi_{32} & \phi_{33} \end{pmatrix} \begin{pmatrix} AY_{i-1} \\ N\mathrm{eff}_{i-1} \\ N\mathrm{dfa}_{i-1} \end{pmatrix} + \begin{pmatrix} \omega_{AYi} \\ \omega_{N\mathrm{eff}_i} \\ \omega_{N\mathrm{dfa}_i} \end{pmatrix}. \tag{9}$$

For alfalfa, yield values and *N*dfa values were assumed to be known only for those locations with closed symbols (Fig. 11.9b).

The model results show that for those locations where ryegrass and alfalfa yield values were ignored (Fig. 11.9b; open symbols), the 95% fiducial limits of estimation increased with distance from the observed location. For both crops, however, the

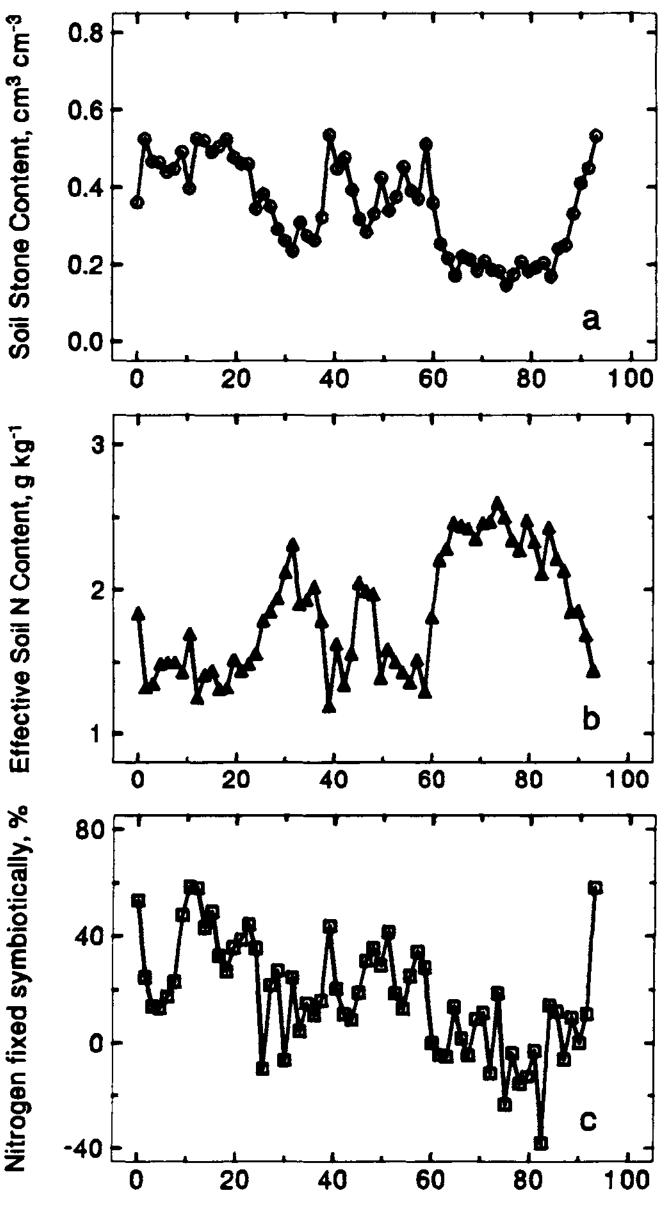

Fig. 11.7. Stone content (a), effective soil nitrogen content *N*eff (b), and the fraction of nitrogen fixed from symbiotic N assimilation *N*dfa (c) in an experimental field in Seibersdorf, Austria.

estimation accuracy of the state-space model was generally sufficient, and the models tended to fail only when large fluctuations occurred between neighboring points. This example also shows that, although many other parameters might have

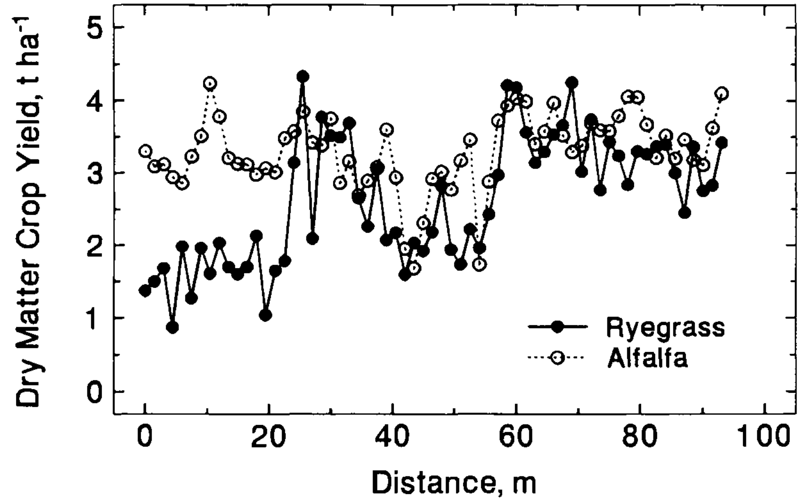

Fig. 11.8. Ryegrass and alfalfa dry matter yield across the transect in an experimental field in Seibersdorf, Austria.

influenced the spatial process of crop yield and symbiotic nitrogen fixation (e.g., soil texture, air-filled porosity, soil temperature, soil water status, oxygen deficiency in the rhizosphere, pH value, organic matter content, micronutrients essential for symbiotically fixing enzymes, etc.), they do not necessarily have to be sampled. The importance of their unsampled contributions in causing deterministic influences on crop growth and yields is integrated into the model error (Nielsen et al., 1994b). Whenever state-space model errors are small, partial information derived quickly from on-site monitored observations can improve our understanding of the field situation and thereby provide a basis for better management decisions. In such situations the quickly observable variables reflect the main underlying process in fields. On the other hand, whenever state-space model errors are unacceptably large, additional or different soil or environmental parameters must be measured or derived either through existing knowledge or deterministic research.

D. Spectral analysis

Knowledge of the spatial pattern of soil properties is necessary for achieving higher efficiency of input for crop production. Such knowledge can also indicate the impact of previous agricultural management practices. Most agricultural field operations occur with a regular pattern. For example, a tractor goes back and forth across the field with an implement in parallel paths at regular intervals. Also, plants are grown with a regular pattern in order to decrease inter-crop competition and to increase water, light, and nutrient use efficiency. Moreover, considering the time domain, many field soils are subject to a certain crop rotation system, repeating in cycles of several years. Hence, periodic patterns develop spatially, temporally, or in

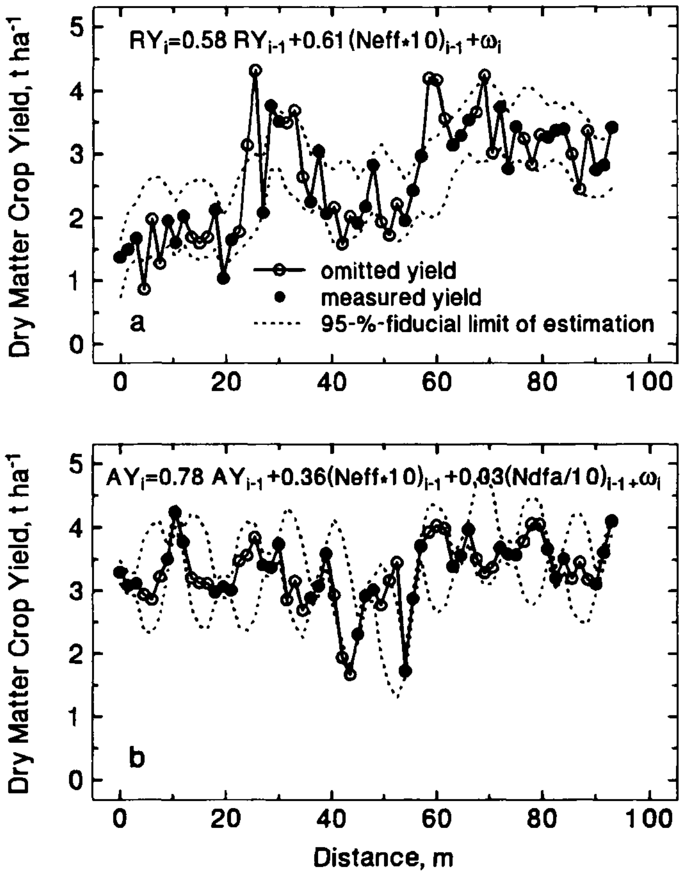

Fig. 11.9. State-space estimation of ryegrass yield (a) and alfalfa yield (b) across the transect in an experimental field in Seibersdorf, Austria.

both domains. Nielsen et al. (1983) showed how variation of soil moisture can be separated into cyclic components, corresponding to management operations. Bazza et al. (1988) applied a procedure known as spectral analysis to soil temperature data to show that patterns in these data coincided with the sinusoidal application pattern of irrigation water with different salt content. Using standard correlation methods, Kachanoski et al. (1985a) found that microtopography and A-horizon parameters were not related. However, when they considered the sampling coordinates of the parameters, and applied cospectral and spectral analysis techniques, it was found that spatio-periodical relations did indeed exist.

A series of observations can manifest various periodic patterns with different amplitudes and different lengths of cycles. The length of a periodic cycle is designated by the wave length (λ) or period, which is the inverse of the frequency (Davis, 1986). As a hypothetical example, three series with wave lengths of 4, 15, and 45 (i.e., one cycle every 4, 15, and 45 length units), respectively, and different amplitudes are drawn in

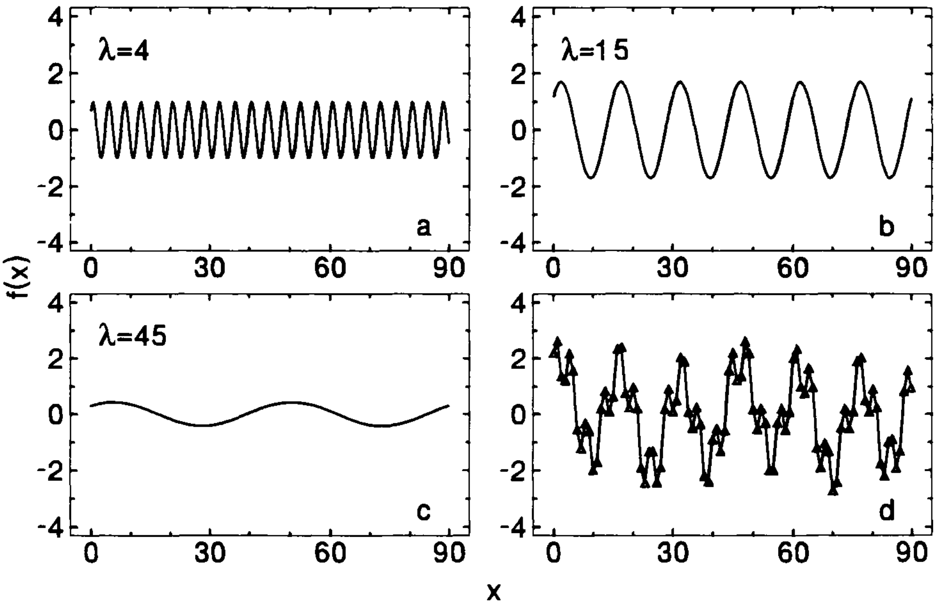

Fig. 11.10. Hypothetical sinusoidal series of different period and amplitude (a, b, c), and the integrated series, "sampled" at 91 locations (d).

Figure 11.10a,b,c. In the literature, short-range variation is often attributed to agricultural management practices (Trangmar et al., 1985, Kachanoski et al., 1985b, Moulin et al., 1994), whereas long-range variation reflects geologic components. When the three patterns are superimposed and sampled at 91 positions with observations separated by one length unit, one gets the confounded pattern in Figure 11.10d.

The power spectrum $f(\lambda)$ of the process x_i as a function of wave length λ is obtained via Fourier transformation by the following:

$$f(\lambda) = \sum_{-\infty}^{\infty} C(h) \exp[-2\pi i \lambda h] \tag{10}$$

where $i^2 = -1$

Spectral analysis filters the periodic variance components, shown in Figure 11.11 for the hypothetical example. One can find the three peaks in the power spectrum at the corresponding frequencies of 1/4, 1/15, and 1/45 (length units)$^{-1}$, respectively. Having determined a cyclic behavior of a series, one may use the knowledge about the period to make forecasts (e.g., forecasting the weather or river levels is a common practice in the meteorological and hydrological sciences; Kite, 1989).

The example here is yields across the two neighboring transects of the almond orchard considered above (Fig. 11.3), which are later used for cokriging (Fig. 11.5). Sprinklers were located after every third tree in the orchard, and the farmer was interested in whether this sprinkler arrangement affected tree growth. If this was so, the effect should have accumulated over the years and be manifested in a periodic

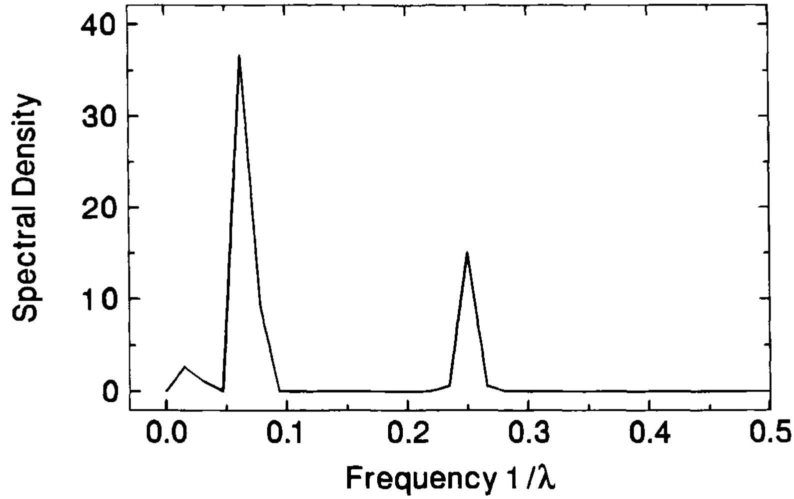

Fig. 11.11. Power spectrum for the hypothetical data set in Fig.11.10d.

variation of some easily obtainable tree growth parameters. For this purpose, trunk circumference of trees within a transect (Fig. 11.12) measured at 50 cm above the soil surface was analysed after the original data had been detrended. A peak appears at a frequency close to 0.33 in the power spectrum (Fig. 11.13) (corresponding to a period of every third tree). That this peak occurs at the same frequency as that of spatial sprinkler distribution indicates that irrigation design does effect tree growth. The increase of the power at $1/\lambda = 0.5$ (length units)$^{-1}$ reflects the fluctuation from one tree to the next and is perhaps due to inter-plant competition.

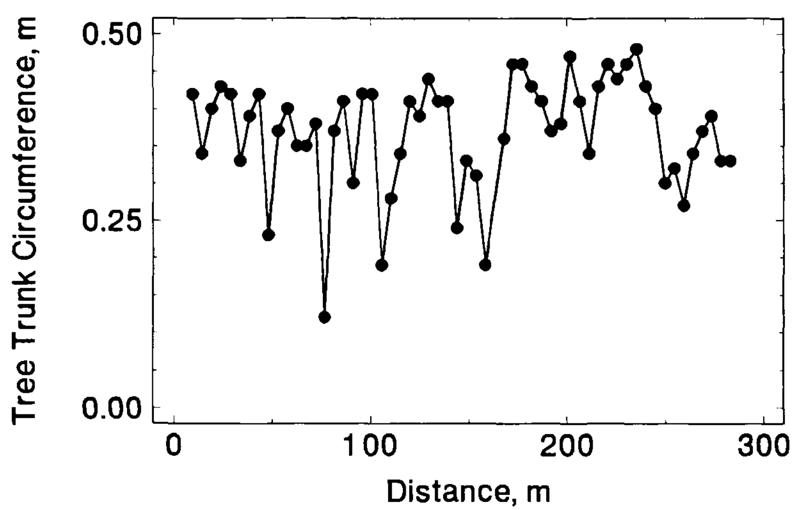

Fig. 11.12. Trunk circumference of almond trees in a transect north of Sacramento, Calif., U.S.A.

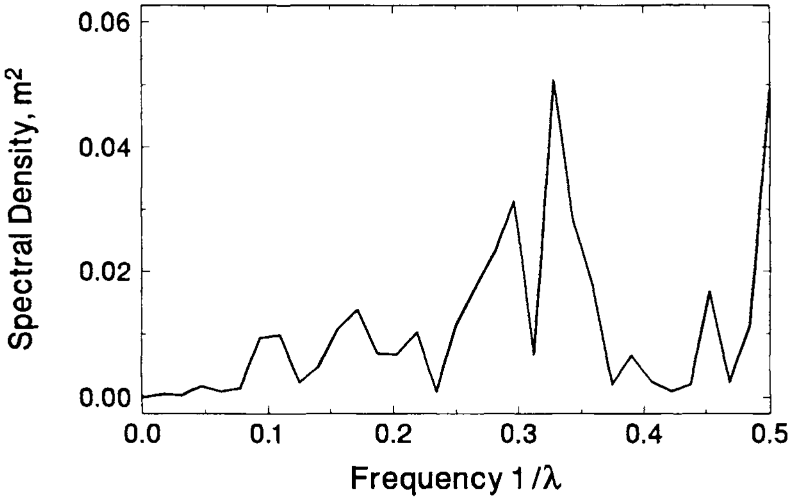

Fig. 11.13. Power spectrum of detrended almond tree trunk circumference in a transect (from Fig. 11.12) north of Sacramento, Calif., U.S.A.

When conducting spectral analysis, one must use samples at regular intervals in space or time. Moreover, the sample frequency should be higher than the expected frequency of the process or pattern being examined. This may cause laborious sampling, but sampling can be undertaken instantaneously and directly without designing any experiment or invoking any treatment in the field. Moreover, simple variables can be examined on-site (e.g., tree trunk circumference) to give direct information about the specific site to the farmer. In the almond orchard example, it would have been a monumental effort to design a field experiment in which the effects of sprinkler position had to be investigated in a randomized block experiment and the assumptions for classical statistics had to be obeyed (i.e., that observations had to be independent of each other). Had a randomized block experiment been conducted, still no information for the farmer's site would have been obtained.

One can also use spectral analysis to determine whether the frequency-dependent variations of two series of observations coincide (i.e., whether they are coherent). For example, the squared coherence function $\kappa(\lambda)$, as a measure of frequency dependent correlation, is determined for the two series of almond yields in parallel transects (Fig 11.3) according to:

$$\kappa_{yx}(\lambda) = \frac{[f_{yx}(\lambda)]^2}{f_x(\lambda) f_y(\lambda)} \tag{11}$$

where $f_{yx}(\lambda)$ is the cross spectrum (Shumway, 1988). The squared coherence for frequency-dependent analysis of variance is analogous to the coefficient of determination in classical regression and has values between 0 and 1. It indicates at which wavelengths two series proceed coherently or coincidentally. Therefore, the squared coherence is 1 at all frequencies if one series x_t is an exact linear filter of

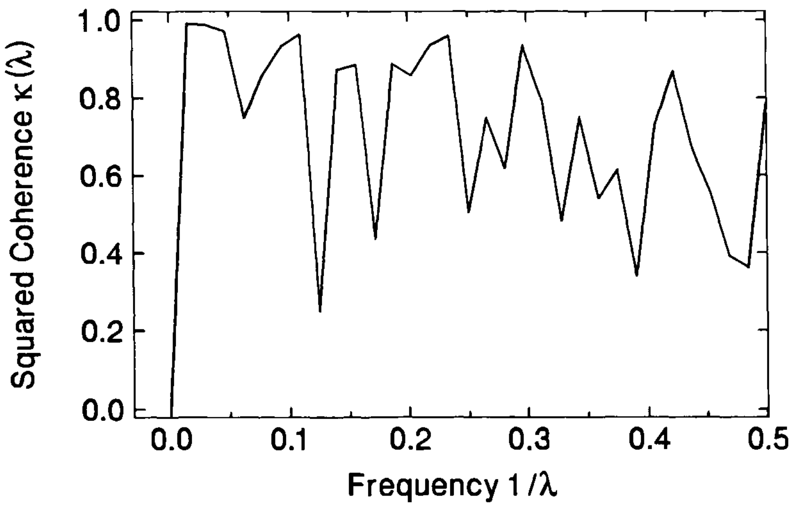

Fig. 11.14. Squared coherence of almond yields in two parallel rows (see Fig. 11.3) in an orchard north of Sacramento, Calif., U.S.A.

another series y_t. The spectrum of squared coherence of the almond yields in the parallel transects (Fig. 11.14) shows a strong coherence for several short wavelengths and especially for long wavelengths, and may indicate similarities in the periodic variation of some underlying soil properties and growing conditions in both transects.

E. Analyzing spatially variable field observations with physically based equations

Thus far in this chapter, we have advocated the use of spatial statistics in addition to the currently used classical methods. In both cases the statistical analyses examine variance and covariance functions derived from observations and measurements without explicitly invoking a physically based equation. The various kinds of observations selected were expected to be correlated based upon a conceptual knowledge of the processes occurring in the field. If correlations were not found, other kinds of observations would have to be selected by trial and error. Here we introduce the idea of using a physically based equation in combination with a set of observations expected to be correlated in space (or time).

A nonlinear partial differential equation describing a physically based process occurring at the soil surface can be derived and transformed into a state-space formulation. The process may be physical, chemical, or biological in nature (e.g., infiltration, nitrification, the leaching of soil solutes in the presence of plant root extraction, etc.). Such state-space models simultaneously examine a theoretical equation, its empirical parameters, and the observations that embrace the uncertainties of soil heterogeneity and instrument calibration. The usefulness of this approach lies in the opportunity to be guided by an equation expressing a process that occurs at the soil surface and to simultaneously analyze the uncertainty in both the equation and our field measurements. Examples of progress recently achieved to improve our assessment of soil quality using state-space approaches include the examination of evaporation (Parlange et al., 1993) and infiltration and redistribution of soil water (Katul et al., 1993; Wendroth et al., 1993).

A desirable feature of the state-space methodology is the inclusion of an observation error that can be treated as a known, measured quantity or, alternatively, as an unknown for which a solution is found in the numerical scheme. The magnitude of a known observation error allows a reconsideration of the state variable in the equation or an improvement in instrumentation or calibration. On the other hand, by treating the observation error as an unknown, its behavior in space and time can be related to spatial and temporal correlation lengths that may manifest themselves within the domain of the field being studied.

As an example for applying physically based equations in the state-space analysis, the soil water transport equation is employed in order to determine the hydraulic conductivity function of a soil layer from time series field observations of soil water content $\theta(t)$ and hydraulic head difference across depth (i.e., the hydraulic gradient, $(\mathrm{d}H\mathrm{d}z^{-1})(t))$ (Fig. 11.15). These series were determined during water redistribution of an internal drainage experiment that was undertaken at the Campbell Tract experimental field of the University of California, Davis. Details of the underlying experiment are given in Katul et al. (1993) and Wendroth et al. (1993).

A soil layer between the upper depth z_i and the lower depth z_{i+1} is considered, for which we want to estimate the hydraulic conductivity–soil water content relationship $K(\theta)$. The soil water storage in this layer is W defined as follows:

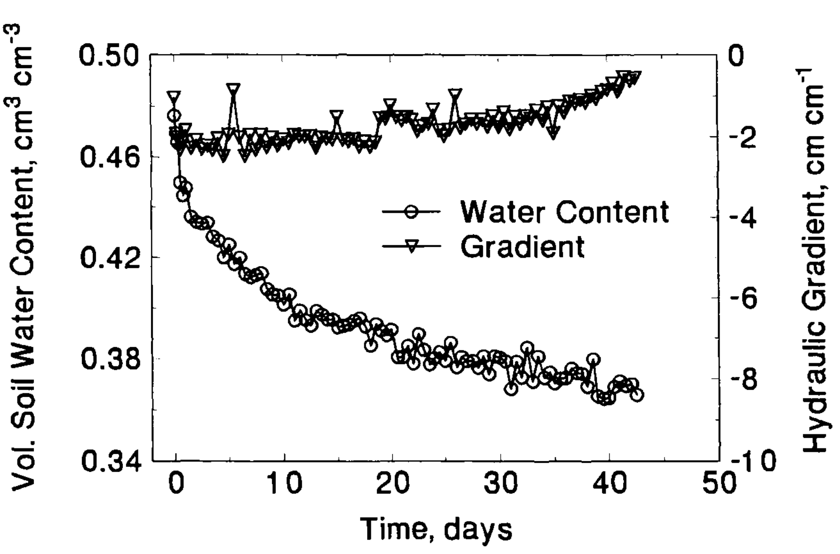

Fig. 11.15. Soil water content and hydraulic gradient time series in the surface soil layer during 43 days of an internal drainage experiment in Davis, Calif., U.S.A.

$$W = \int_{z_i}^{z_{i+1}} \theta \, dz \tag{12}$$

In the experiment, the plot was covered with a plastic sheet to prevent any soil water flux at the upper boundary (q_i), hence the upper boundary condition is zero flux. The water storage change in time of the upper soil layer that we are interested in has to be attributed to drainage flux (q_{i+1}) at depth z_{i+1}. The water storage change in time is then:

$$\frac{\partial W}{\partial t} = -q_{i+1} + q_i \ . \tag{13}$$

The force that is driving the flux q_{i+1} (i.e., the hydraulic head H difference across depth) is measured at the center between z_i and z_{i+1} and below z_{i+1}. Using Darcy's law, Equation 13 can be written as follows:

$$\frac{\partial W}{\partial t} = -K(\theta)\frac{dH}{dz} + q_i \tag{14}$$

Note, that $K(\theta)$ is the function in which we are interested. This function reflects highly relevant soil pore system properties influenced by soil type, land use and management. It is often used in equations for irrigation control, water budget modelling, forecasts, etc.. The following simple two-parameter exponential function is employed:

$$K(\theta) = A \exp(BW) \ . \tag{15}$$

Combining Equations 14 and 15 yields

$$\frac{\partial W}{\partial t} = -A \exp(BW)\frac{dH}{dz} + q_i \ . \tag{16}$$

In order to formulate a state-space equation, soil water storage in the depth compartment is considered as the state variable $X(t)$. Moreover, model errors $\omega(t)$ are included and can be addressed to misleading assumptions underlying Equations 14 and 15. The state-space equation is then:

$$\frac{dX(t)}{dt} = -A \exp(BX(t))\frac{dH}{dz}(t) + q_i(t) + \omega(t) \ . \tag{17}$$

Inasmuch as the true state of soil water storage in the soil compartment cannot be determined but is estimated indirectly with a neutron probe, an observation equation has to be defined as follows:

$$Z(t_k) = X(t_k) + ny(t_k), \quad k = 0, 1, 2, 3, \ldots \tag{18}$$

The noise term $ny(t_k)$ accounts for instrumental calibration and measurement errors of the neutron probe.

In the state-space analysis, an expectation for Equation 17 and the variance being the squared difference between state and state expectation are calculated (see Katul et al., 1993). The propagation of the state and its variance need to be solved simultaneously. Initial estimates of $K(\theta)$ model parameters A and B, and of some

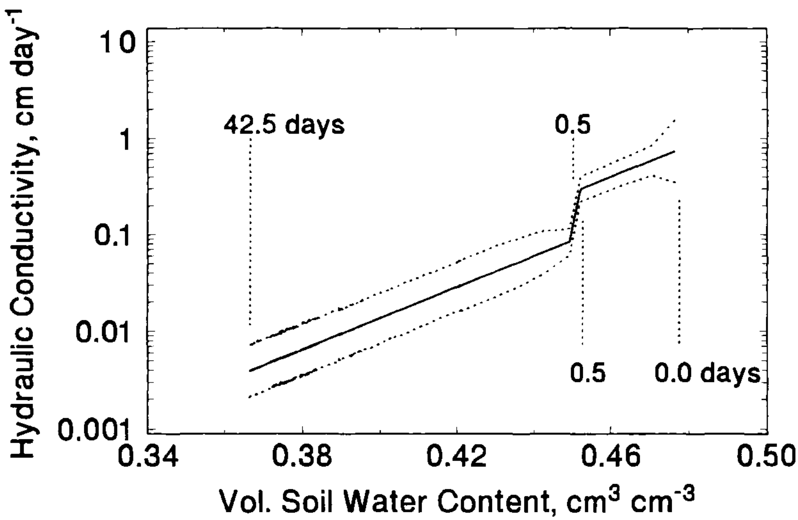

Fig. 11.16. Hydraulic conductivity as a function of soil water content determined from the internal water drainage in the surface soil layer (from Fig. 11.15) in Davis, Calif., U.S.A.

initial conditions are given. These are iteratively optimized via prediction, comparison between the prediction and the observation, and updating of each time step with respect to the variance. At this point the more interested reader is referred to Gelb (1974), Katul et al. (1993) and Wendroth et al. (1993).

The resulting $K(\theta)$ relation is shown in Figure 11.16. In order to achieve an appropriate prediction of $\theta(t)$, the water content time series was divided into two domains, one observed during the first 12 hours of the experiment, the other afterwards, probably manifesting transport phenomena and properties at different pore domains. In the range of high water contents, mainly macropores contribute to water transport, whereas $K(\theta)$ apparently follows a different relation in the drier range. For further details, see Wendroth et al. (1993).

This application of state-space analysis and Kalman filtering seems to be similar to an inverse estimation procedure, such as that of Kool and Parker (1987), where a transport model equation repeatedly runs in combination with a nonlinear optimization routine until a convergence criterion for a set of empirical parameters is met (i.e., the objective function is optimized). Nevertheless, there exist distinct differences between the inverse procedure and the state-space analysis. These differences are the same as those mentioned earlier when comparing estimation of autoregression coefficients in a classical regression analysis versus state-coefficient estimation in the Kalman filtering procedure. Neither the classical regression nor the inverse nonlinear optimization of an empirical relation in combination with a physically based equation account for measurement and model error, nor do they imply an updating within the range of possible variance resulting from measurement and model uncertainties. Instead of taking advantage of observations during prediction of a series wherever they become available, as it is done via updating in the Kalman filter, inverse procedures compare observations and estimations only within the objective function at the end of an iteration step. Unlike Kalman filtering, they do not incorporate any quantity of model error nor yield any measurement error, respectively.

F. Combining water and solute transport models with geographic information systems

Similar to the state-space approach presented above, the following example also takes advantage of physically based equations. Reynolds et al. (1994, 1995) used a mechanistic water and solute transport model in combination with georeferenced soil, weather and crop management data to estimate the potential for leaching of the herbicide, atrazine, into the ground water under an entire watershed. The model, which was a modified form of the modelling package LEACHM (Hutson and Wagenet, 1989) integrated the major processes that occur in the soil profile, including soil horizonation; saturated, unsaturated, steady and transient water flow; crop management, growth, and transpiration; solute sorption, degradation, advection, and dispersion; precipitation and evaporation; soil heat flow; and water table elevation. Soil survey information was used to obtain the required model input data on soil properties. Archived weather data records were used to derive the necessary model input for weather. Crop management practices for a corn (*Zea mays* L.) crop were assumed, with planting, harvesting, and atrazine application dates being determined by both soil properties and weather. All input data were georeferenced to the centroids of 119 soil landscape polygons that encompassed the watershed of interest. The model was run at each of the 119 landscape polygon centroids for a period of 10 consecutive model years; and model predictions of 1) annual atrazine loading at the 90-cm (average tile drain) depth, and 2) elapsed time for atrazine to reach the 3-ppb concentration (U.S. EPA drinking water limit) at the 90-cm depth were collected in space and time. Kriging was then used to convert the 119 irregularly spaced and highly variable point values of soil properties, weather data, and predicted atrazine loadings and concentrations into 1657 interpolated values extending throughout the watershed on a regular 2 km by 2 km grid. The kriged interpolations accomplished the required extension from a point basis (polygon centroids) to an areal basis (watershed), while still retaining the spatial variability characteristics of the original data. The kriged data also provided the spatial detail necessary to allow a Geographic Information System (GIS) to effectively produce and overlay maps of atrazine loading and concentration, soil properties, and weather data.

The predicted atrazine loadings to the 90-cm depth were found to be highly variable and complexly distributed throughout the watershed (Fig. 11.17). Comparison of this loading map to soil texture and summer precipitation maps for the watershed (texture and precipitation maps not shown) revealed that the lowest atrazine loadings occurred where soils were clayey and summer precipitation was low, whereas intermediate to high loadings occurred on sandy to loamy soils where summer precipitation was moderate to high. Correlation analysis showed further that atrazine loading was significantly correlated with many soil and weather

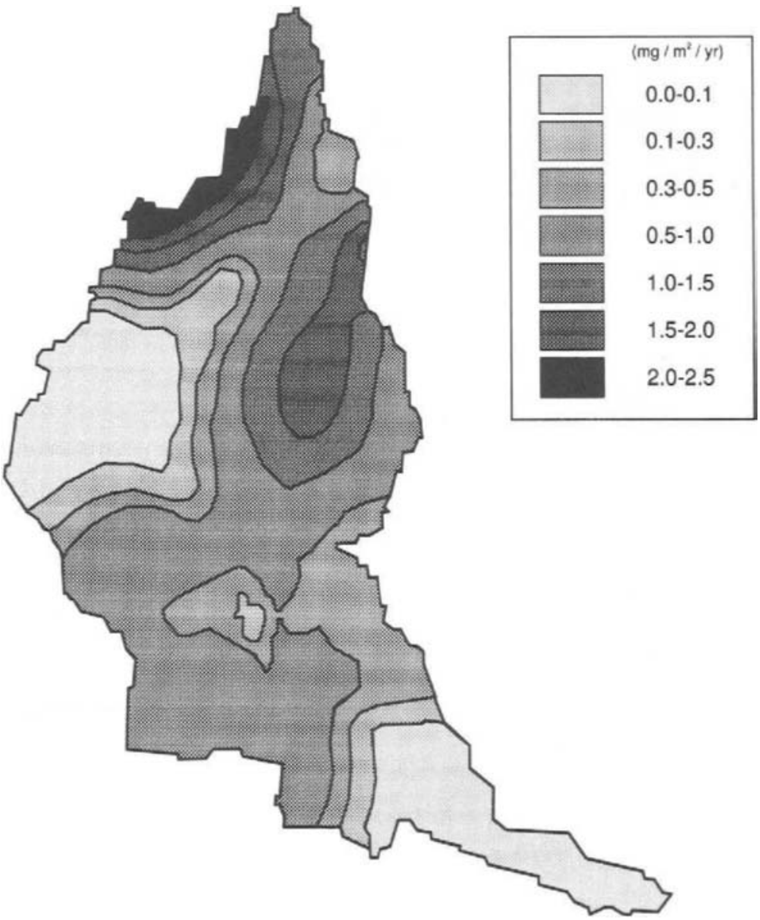

Fig. 11.17. Annual atrazine loadings (mg atrazine m^{-2} yr^{-1}) at the 90-cm depth for the Grand River watershed, Ontario, Canada.

parameters, but the magnitudes of these correlations were generally low. This suggests that atrazine loading in the watershed was determined by complex interactions among several soil, weather, crop management, and solute transport factors, rather than by one or two dominant factors.

The concentrations of atrazine in the soil water at the 90-cm depth were predicted to be generally low throughout the watershed (Fig. 11.18). The 3 ppb U.S. EPA drinking water guideline for atrazine was exceeded, however, on or before the tenth simulation year in about 27% of the watershed area (Fig. 11.18). The areas where this occurred also have predicted annual atrazine loadings that fall within the top half of the loading range (i.e., 0.5–2.5 mg atrazine m^{-2} $year^{-1}$, Fig. 11.17), which may consequently suggest that Figure 11.18 demarks regions of potentially significant low-level non-point source contamination of groundwater by downward migration

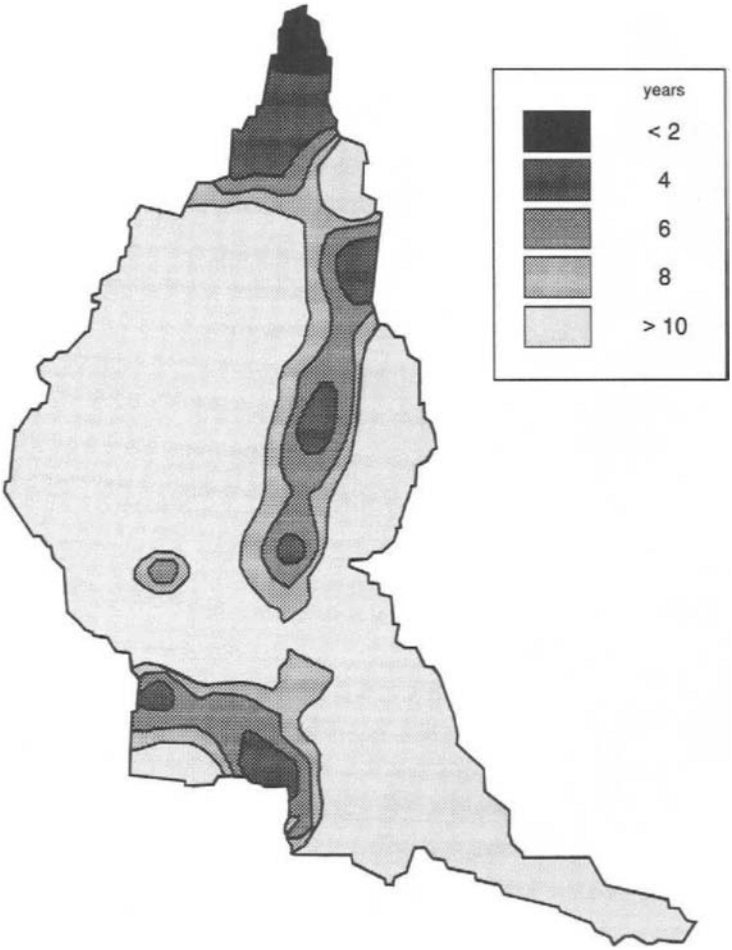

Fig. 11.18. Predicted time for atrazine to reach the 3-ppb concentration at the 90-cm soil depth in the Grand River watershed, Ontario, Canada.

of atrazine through the soil profile. Further more detailed investigations may therefore be warranted in the areas where atrazine concentrations are predicted to be above the U.S. EPA limit.

Although combined simulation model–geostatistics–GIS analyses are still in the preliminary stages of development, it is clear from the above example that such analyses are potentially very powerful and useful. These analyses could potentially be used to determine the importance and spatial-temporal distributions of a process (e.g., pesticide leaching); to determine the major soil, land use, and environmental factors controlling the process; to estimate the potential environmental impact of changes in land use and land management; and to help establish land use and land management practices that are both optimal and environmentally sustainable.

III. CONCLUSIONS

As a compendium of approaches described in this chapter, Table 11.1 gives an overview of the main statistical approaches currently being used in soil science. As Table 11.1 represents only a small proportion of what is possible, many new statistical approaches will undoubtedly be applied in the future.

TABLE 11.1

Compendium of approaches for assessing soil quality based on spatial and temporal statistics

Tool	Purpose	References
Autocorrelation Function	Plot of the correlation of a variable with itself across a distance h (lag); reflects the spatial or temporal continuum; assumed to be zero in ANOVA, regardless of the distance between observations	Shumway (1988), Morkoc et al. (1985b), Isaaks and Srivastava (1989)
Semivariogram (Variogram)	Plot of half of the average squared difference between observations separated by a distance h; like a mirror image of the autocorrelation function; reflects the range over which observations are correlated; parameterized by various models with nugget, range, and sill for interpolation (applied in kriging, and cokriging)	Vieira et al. (1981), Trangmar et al. (1985), Davis (1986), Nielsen and Alemi (1989),
Kriging	Spatial interpolation; estimation of values for unsampled locations, based on values at neighboring locations and the spatial (or temporal) variability structure (manifested by the variogram); estimation of confidence bands for interpolated value; jack-knifing is a special kind of kriging for validation of a variogram model	Vieira et al. (1981), Warrick et al. (1986), Alemi et al. (1988),
Crosscorrelation Function	Plot of the correlation between two variables as a function of their separation distance (lag); reflects the distance over which one variable is correlated with the other; length of crosscorrelation reflects the distance over which it is valid to correlate one variable with another	Nielsen et al. (1983), Davis (1986), Shumway (1988)
Crossvariogram (Covariogram)	Reflects the range over which observations of one variable are related to another; parameterized in the same manner as a variogram; used input for (applied in cokriging)	Alemi et al. (1988), Kachanoski and De Jong (1988), Zhang et al. (1995)

Table 11.1 (continued)

Tool	Purpose	References
Cokriging	Multivariate interpolation of values at unsampled locations; often applied to estimate expensive variables (sampled at low density) based on the spatial (or temporal) pattern of a cheap variable (sampled at high density); estimation of confidence bands based on the variograms and the crossvariogram	Alemi et al. (1988), Deutsch and Journel (1992), Smith et al. (1993), Zhang et al. (1995), Halvorson et al. (1995)
State-Space Analysis	Special autoregressive approach; reflects the relation between the state of one or several variables to the state at previous locations (or times); spatial interpolation of unsampled locations; unlike kriging, not limited to stationarity assumptions; accounts for measurement and model uncertainty; incorporates as much deterministic input as necessary and integrates un sampled information on the basis of relations between neighboring observations.	Morkoc et al. (1985a), Nielsen and Alemi (1989), Wendroth et al. (1992), Parlange et al. (1993), Katul et al. (1993), Wendroth et al. (1993), Nielsen et al. (1994a)
Power Spectrum	Decomposing the variation or fluctuation of a series of observations, which is sampled at regular intervals, into periodical components; reflects amplitude and frequency regardless of phase shift; often used to predict hydrological time series; detects effects due to the regular pattern of agricultural operations	Nielsen et al. (1983), Davis (1986), Kachanoski et al. (1985a), Bazza et al. (1988), Shumway (1988)
Coherency	Reflects at which wave lengths or periodicities two series fluctuate coincidently, regardless of any phase shift between the two series; equals 1, if a series is linear filter of another series; analogue to the coefficient of determination	Nielsen et al. (1983), Bazza et al. (1988), Shumway (1988)

The above examples were intended to give some insight into opportunities for assessing soil quality using spatial and temporal statistical approaches. Overall, a number of statistical tools are available for sampling and analysing spatial and temporal processes in ecosystems and agricultural landscapes. Nevertheless, there are no unique answers to questions regarding appropriate sampling schemes and sampling scales. Most investigators commonly use sampling schemes consistent with deterministic concepts applied to small areas or volumes at a specific location.

Although mindful of the much larger dimensions of the domain, plot, field, or agricultural landscape across which the observation or measurement will be interpolated or extrapolated, most investigators think of only one scale—that which is most convenient for the parameterization of the deterministic soil or crop property at a specific location. From the information presented in this chapter, it is obvious that the investigator needs to consider two different scales of observation. The first is the small scale associated with the minimum distance between pairs of observations below which interpolation of values can be neglected. The second is the large scale associated with the maximum distance between pairs of observations above which extrapolation of values can also be neglected. As an example for the consideration of crop production in a farmer's field, the minimum distance might be the distance between individual plants while the largest distance would be the length of the entire field managed in the same manner. Hence, after choosing the kinds of measurements or parameters to be observed, the researcher must decide upon the minimum distance to take observations and an adequate sampling method to achieve a spatial (or temporal) continuum within the entire, larger domain.

Vieira et al. (1981) recommended a spatial density of samples just necessary to detect the spatial continuum and to take additional samples separated by shorter distances in order to improve the estimation of a semivariogram close to the origin for decreasing the estimation variance with spatial interpolation. This improvement can be achieved with so-called nested sampling. One still has to keep in mind that spatial structure may vary between different variables, such as crop yield and soil parameters (Warrick and Gardner, 1983). Moreover, spatial structure changes with time, especially for agronomically relevant variables such as NO_3-N content (Cahn et al., 1994). On the other hand, Or and Hanks (1992) found similar spatial structures for soil water, crop height, crop yield variability, and irrigation water.

The approaches presented here can be easily expanded to different scenarios of on-site and landscape sampling in the space and time domain. They do not give answers to every question, and sometimes they fail. On the other hand, applying spatial statistics on data from field experiments which were originally designed for ANOVA is usually inappropriate or inadequate. Agricultural designs for ANOVA require that observations between treatments be spatially and temporally independent and if observations are not found to be independent in the ANOVA design, there are usually too few observations to make reliable conclusions using spatially dependent concepts.

If spatially variable concepts and appropriate statistical analyses are initially considered in the design of assessing soil quality, the kinds of questions and breadth of answers achievable are more comprehensive and much more flexible than those limited to the classical ANOVA traditionally used in soil and agronomic sciences.

Imposing treatments and looking for an average behavior of a certain kind of treatment on a soil assumed to be homogeneous on the average can be avoided with spatial statistics. Instead, spatial statistics allows direct sampling and analysis of field information for the benefit of resources management. Moreover, with noise or variance components (model and error) being accounted for within prescribed fiducial limits, the results are often much more relevant than average values.

Around the world, the request upon scientists focuses increasingly on the relevance of research for field and landscape scales, and the benefit of their work is judged on the welfare of the environment and society. The statistical analytical tools presented in this chapter combined with integrative indices of soil quality allow for direct on-site analysis and can be considered as one of several important steps in updating our landscape–ecological research strategies to achieve sustainable crop production and maintain optimum ecosystem health.

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Chapter 12

SOIL ORGANIC MATTER DYNAMICS AND THEIR RELATIONSHIP TO SOIL QUALITY

H.H. JANZEN, C.A. CAMPBELL, B.H. ELLERT and E. BREMER

I. INTRODUCTION

Soil is dynamic, always evolving in response to its environment (Elliott et al., 1994). Change occurs in numerous constituents, each of which adapt over different time scales: the soil solution may change within seconds, microbial populations within days, and mineralogy over centuries or millennia. These responses, individually and interactively, affect soil health or quality.

In this chapter, we focus specifically on temporal changes in soil organic matter (SOM). Organic matter merits attention, because it is widely perceived as an indicator of soil productivity (Johnston, 1991). Furthermore, SOM is mutable and very responsive to external influences; many indices of soil quality are fixed, but SOM can be altered, particularly in agroecosystems. Our objectives are to provide a conceptual view of SOM change and infer some implications of such change for soil quality. These objectives are addressed using examples drawn largely from studies on the semi-arid prairie of western Canada.

II. OVERVIEW OF THE CARBON CYCLE

Carbon enters the soil via photosynthesis, which converts atmospheric CO_2 into organic compounds that eventually find their way into soil as plant litter, roots, and root exudates (Fig. 12.1). These organic residues are decomposed by soil fauna and microorganisms, resulting in the subsequent release of much of the C to the atmosphere as CO_2; typically, about 70% reverts to CO_2 within the first year

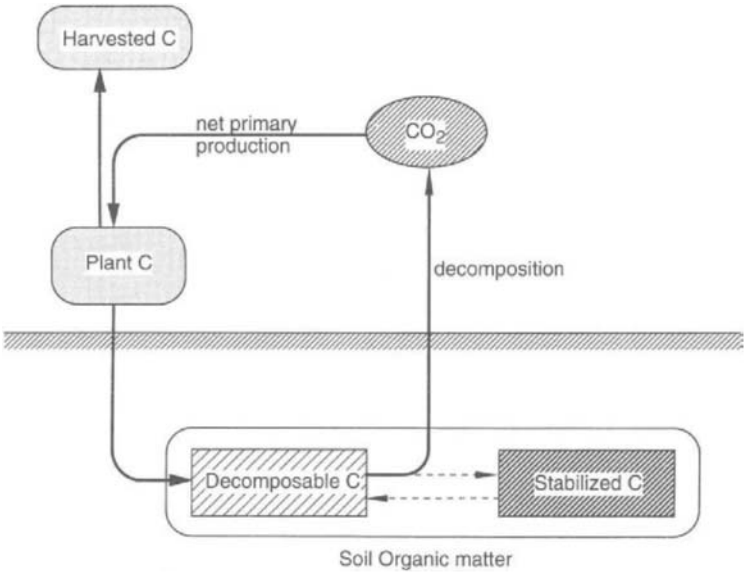

Fig. 12.1. Simplified overview of the C cycle in an agroecosystem.

(Jenkinson et al., 1991). The added C remaining, including that assimilated into microbial biomass, undergoes further decomposition at a slower rate. A small portion may become inaccessible to biological decay by virtue of its chemical composition or by association with soil minerals.

As a result of the ongoing process of C additions and decomposition, SOM includes a continuum of materials ranging from the highly decomposable to the very recalcitrant. This continuum is often divided somewhat arbitrarily into two pools: inert organic matter and decomposable organic matter (Hsieh, 1992, 1993).

The inert organic matter is highly resistant to biological oxidation because of its molecular structure or physical protection, and may have turnover times measured in thousands of years (Campbell et al., 1967; Harrison et al., 1993; Scharpenseel and Becker-Heidmann, 1994). As a result, this fraction shows little change over ecological time periods and is virtually unaffected by management practices. In many soils, as much as 60% of the organic matter may be effectively resistant to biological decomposition (Oades, 1989; Wagner, 1991; Buyanovsky et al., 1994; Nicolardot et al., 1994).

The second pool, the labile or decomposable organic matter, comprises material in transition from fresh residues to CO_2 or inert C. Much of it consists of recently incorporated plant, faunal, and microbial debris, with a turnover time of less than a decade (Trumbore, 1993; Buyanovsky et al., 1994). This pool has been variously described, depending on the approach used in its estimation; specific terms include "light fraction" organic matter (Greenland and Ford, 1964), particulate organic matter (Cambardella and Elliott, 1992), macro-organic matter (Gregorich and Ellert, 1993), mineralizable C (Campbell, 1978), coarse organic matter (Tiessen et al., 1994), and organic matter in macroaggregates (Buyanovsky et al., 1994). Although it is unlikely that any of these techniques cleanly extract the decomposable pool, all provide a reliable measure of relative changes in transitory SOM.

III. CHANGES IN SOIL ORGANIC MATTER

A. A conceptual view

Changes in SOM occur whenever the rates of C input (net primary production) and C loss (decomposition) diverge. Any force that disproportionately affects primary production and decomposition will elicit a change in SOM content.

The organic matter present in uncultivated soils accumulated over many centuries, a process succinctly summarized by the concept of succession (Odum, 1969; Schlesinger, 1991; Johnson, 1995). During early stages of ecosystem development, primary production exceeds respiration, resulting in the accumulation of C in the ecosystem (Fig. 12.2). As the ecosystem matures, however, the rate of respiration eventually converges with the rate of production and, as a result, the rate of C storage approaches zero. At this equilibrium, the amount of stored C represents the integrated difference (area) between the respiration and production curves (Fig. 12.2).

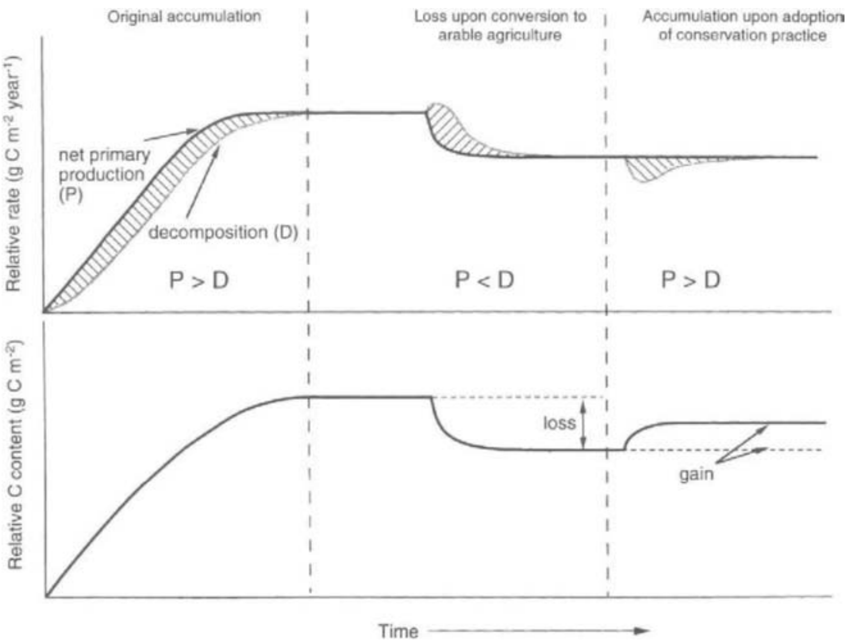

Fig. 12.2. Conceptual view of organic C dynamics in an agroecosystem on grassland soil. In accordance with the succession theory (Odum 1969), net primary production (P) initially exceeds decomposition (D), resulting in the accumulation of soil C until P and D converge. Upon conversion of the land to arable agriculture, D initially exceeds P, resulting in the loss of soil C until a new steady state is approached. Adoption of C-retentive cropping practices reduces D relative to P, resulting in a gain of C until D and P again converge.

In grassland soils, in which C in phytomass is relatively small, virtually all of the accumulated C is stored in soil organic matter.

The imposition of arable agriculture almost inevitably prompts the divergence of respiration and production rates, thereby disrupting the quasi-steady state established in previous centuries. Almost invariably, cultivation enhances respiration relative to C input, resulting in the net loss of stored C in soil (Fig. 12.2). This loss continues until rates of respiration and primary production again converge and a new steady state is approached.

The OM content of cultivated soils (surface layer) is typically about 15 to 30% lower than that of soils under native vegetation (McGill et al., 1988; Anderson, 1995; Gregorich et al., 1995; Ellert and Gregorich 1996). Highest rates of loss usually occur shortly after disruption, and the rate of SOM decline abates thereafter (Campbell, 1978; Tiessen and Stewart, 1983; Bowman et al., 1990; Monreal and Janzen, 1993). Most agricultural soils have now been cultivated long enough to approach a new steady state (Cole et al., 1993; Paustian et al., 1997).

Adoption of new agronomic practices, however, may again disrupt the production–respiration balance, thereby altering the SOM steady state. Modification of variables, such as by nutrient amendment, tillage, or crop rotation, essentially reinitiates the succession process. If the new agronomic practice reduces C input relative to decomposition, then SOM declines. For example, at a site in southern Alberta, establishing cropping systems with frequent summerfallow resulted in SOM decline relative to that under continuous wheat (Bremer et al., 1995). Much of the loss however, occurred within a few decades (Fig. 12.3). Thus SOM loss upon adoption of a degradative practice appears to follow an asymptotic decline similar to that observed following cultivation of native grassland.

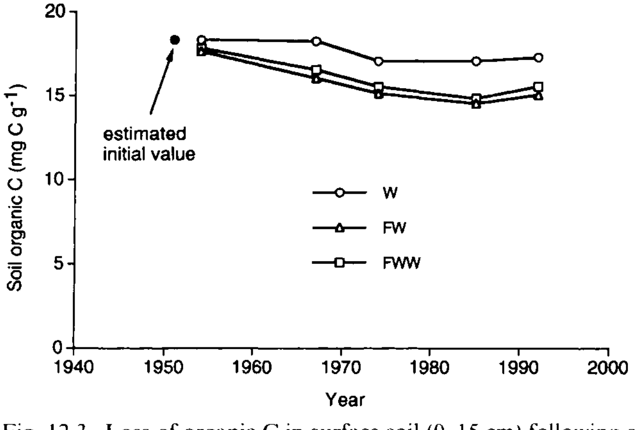

Fig. 12.3. Loss of organic C in surface soil (0–15 cm) following change in management after about 40 years of cultivation. Treatment designations are as follows: W = continuous wheat, FW = fallow–wheat, and FWW = fallow–wheat–wheat. Wheat was unfertilized for most of the study, but phosphorus was applied to cropped phases from 1985 on. Values for FW and FWW are averages across rotation phases. Organic C concentration for 1951 was not measured, but assumed to be the same as that for W in 1954 (adapted from Bremer et al., 1995).

A SOM steady state can, however, also be disrupted positively by adopting a practice that favors C input relative to decomposition (Fig. 12.2). One approach that has received particular attention in recent years is the reduction of tillage intensity. For example, Campbell et al. (1995) measured the SOM response to adoption of reduced tillage in a soil that had previously been under a tilled fallow–wheat system for 70 to 80 years (Fig. 12.4). In combination with continuous cropping and enhanced fertilization, the reduced tillage increased the organic matter content of the 0- to 15-cm soil layer by several Mg C ha^{-1}, relative to an estimate of C content at the beginning of the study. Although such estimates of short-term change have some uncertainty, much of the increase in SOM apparently occurred in the first several years following adoption of the improved practices, and C content appeared to reach a new plateau within about five years. Other studies have also shown that the most rapid increase in SOM occurs within a decade after adopting reduced tillage (Paustian et al., 1997). Accumulation of SOM therefore may follow an asymptotic pattern inverse to that of SOM loss.

The magnitude of the SOM response clearly depends not only on the eventual steady state of the new practice, but also on the C status resulting from previous management. For example, a soil managed using C-retaining practices may have less potential for SOM increases than one under degradative management.

In many agricultural ecosystems, SOM content may rarely attain steady state. Because of evolving management practices, SOM may be in almost continual transition from one trajectory to another.

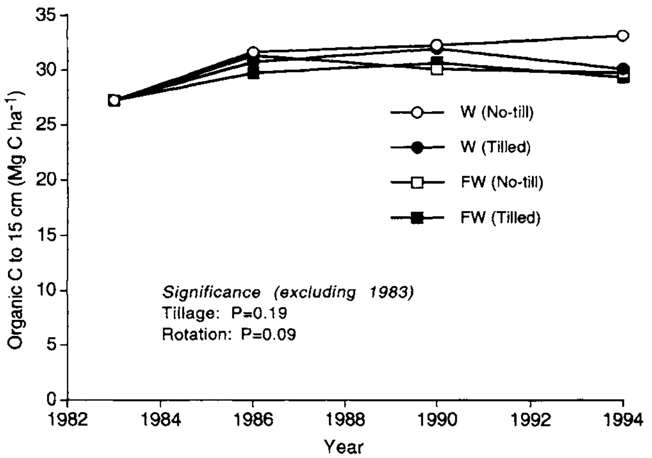

Fig. 12.4. Accumulation of organic C in surface 15 cm of a loam soil after adoption of fertilized spring wheat (W) production systems with different tillage systems (No-till, tilled). Prior to 1983, the land had been farmed for about 75 years in a tilled fallow–wheat (FW) system. The value reported for 1983 is the mean of 12 samples from across the site (standard deviation = 2.3 Mg ha^{-1}); for assumptions, please refer to original source (from Campbell et al., 1995).

B. Mechanism of change

How does introduction of a new practice effect SOM change? One way is by changing the amount of C entering the soil. The SOM loss upon initial cultivation, for example, can be attributed in part to reduced C inputs. Voroney et al. (1981) estimated that annual inputs of C in a Black Chernozem are about 3.1 Mg ha^{-1} in native prairie compared to less than 2 Mg ha^{-1} in various wheat cropping systems. Much of this difference is attributable to the removal from agroecosystems of C in harvested crop materials; for example, about one-third of the C assimilated by wheat is removed in the form of grain (Campbell et al., 1991). Since the objective of agriculture is export of organic products from the ecosystem, arable soils almost invariably have lower C returns than those in undisturbed systems. Other agronomic variables that can influence SOM content by modifying C inputs include fertilizer application, residue removal, crop species and rotation, and addition of organic amendments.

Although adjustment of C inputs is a primary cause of SOM change, altering the decomposition rate may also be important. Organic matter includes a continuum of materials ranging from "raw" plant litter to very stable, essentially inert humus. The transitory material in this continuum is the decomposable or labile fraction (Fig. 12.1). Any factor that impedes the flow of C from plant litter to CO_2 (or inert C) results in the accumulation of decomposable C. Conversely, any factor that eases constraints on C flow results in the depletion of decomposable organic matter. The size of the transitory pool is therefore highly sensitive to the rate of decomposition, and its fluctuations can explain many of the organic matter changes observed upon disruption of equilibria.

One example of SOM decline resulting partly from removal of constraints to decomposition is the loss observed upon initial cultivation of grassland soils. In undisturbed grassland, soils are usually desiccated because of continuous water extraction by vegetation. Because of the sensitivity of soil respiration to water stress (Wildung et al., 1975; Norman et al., 1992), decomposition rates are retarded, resulting in the accumulation of decomposable C. The introduction of annual crops, with their lower and more sporadic water demands compared to native species (de Jong and MacDonald, 1975), provides greater biological opportunity for decomposition, resulting in the depletion of decomposable SOM. For example, Buyanovsky et al. (1987) observed faster decomposition of litter under winter wheat than under native grass. The loss of SOM upon cultivation of grassland soils may be attributed less to the direct physical effects of the tillage than to the stimulation of biological activity via the removal of hydrothermal constraints.

The divergence of SOM in treatments with varying frequency of summerfallow may also be partially attributable to differences in decomposition rate induced by variable moisture constraints. In continuously cropped systems the plants desiccate the surface soil, thereby slowing decomposition (Jenkinson, 1977; Shields and Paul, 1973; Voroney et al., 1989) and resulting in the accumulation of decomposable C. Conversely, the summerfallow period provides moist soil conditions during the period of most favorable temperatures, resulting in the depletion of decomposable C.

Labile organic matter, as a proportion of the total, is therefore usually much lower in systems with summerfallow (Bremer et al., 1994; Biederbeck et al., 1994; Janzen et al., 1992a). Indeed, the loss of decomposable organic matter in the fallow period may account for a large part of the difference in SOM between continuously cropped and frequently fallowed soils.

C. Change in composition

Much of the SOM change in response to management practices occurs in the labile pools. Thus, conversion of soils to agriculture results in the disproportionate loss of decomposable fractions (Schlesinger, 1991; Arrouays and Pelissier, 1994; Cambardella and Elliott, 1992; Harrison et al., 1993). Consequently, decomposable C typically comprises a much lower proportion of total SOM under arable agriculture than in comparatively undisturbed ecosystems. For example, light fraction C is often lower in cultivated soils than under native vegetation (Dalal and Mayer, 1986a; Balesdent et al., 1988). These findings corroborate the suggestion that much of the loss in SOM upon cultivation is attributable to removal of abiotic constraints to decomposition.

Similarly, SOM change in response to altered cropping practices may also occur predominantly in decomposable fractions. For example, Bremer et al. (1995) observed that the light fraction SOM declined to a much greater extent than did total SOM upon introducing summerfallow (Fig. 12.5). As a result of the disproportionate changes in labile fractions, the quality of SOM is often more affected by management

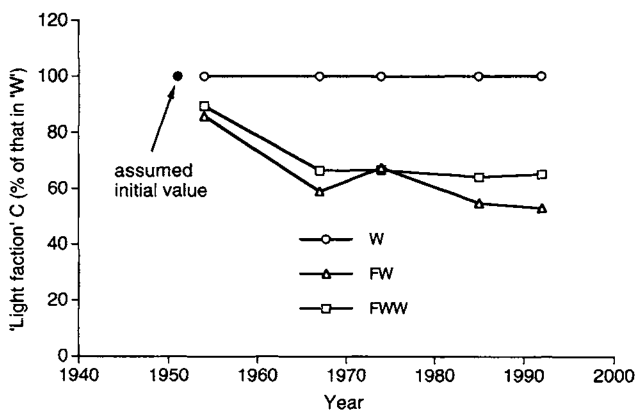

Fig. 12.5. Change in relative "light fraction" organic C following adoption of various spring wheat–fallow rotations. "Light fraction" organic C concentrations are expressed as a percentage of those in the continuous wheat treatment, which fluctuated somewhat among years, perhaps because of variation in sampling procedures (see reference for actual values). Treatment designations are as follows: W = continuous wheat, FW = fallow–wheat, and FWW = fallow–wheat–wheat. Wheat was unfertilized for most of the study, but phosphorus was applied to cropped phases from 1985 on. Values for FW and FWW are averages across rotation phases. The value for 1951 was assumed. (from Bremer et al., 1995)

change than is total organic matter content (Campbell and Souster, 1982; Biederbeck et al., 1994).

Because much of the change in SOM occurs in fractions with short turnover times, SOM changes can occur relatively quickly; adoption of revised cropping systems can often measurably benefit SOM content within a few years (e.g., Angers, 1992). The sensitivity of SOM to revised cropping practices, however, also makes it susceptible to rapid loss if the improved practices are not maintained. If a gain in SOM occurs primarily by constraints to decomposition, then any reversion to conditions more favorable to microbial activity may induce rapid depletion of the accumulated C. These observations suggest that SOM change may be highly reversible.

D. Limits to change

Rates of SOM change upon disruption of a steady state generally decline with time (Dalal and Mayer, 1986b). Highest rates of change are typically observed within the first few years following disruption of the previous steady state, and the rate of accumulation or loss then declines as rates of respiration and production converge. This convergence is assured, because decomposition follows first-order kinetics, whereas the annual C input is better described by zero-order kinetics. The asymptotic pattern of organic matter loss or gain over time may result in the overestimation of the rate of change. As stated by Schlesinger (1990), "*at any time, the long-term rate of accumulation is an overestimate of the current rate.*" Consequently, linear extrapolation of rates estimated from short-term studies may overestimate long-term potential for SOM gain or loss.

The asymptotic pattern of SOM change implies a certain resistance of SOM to change; although SOM is responsive to management, there appear to be rigid limits to the magnitude of change. The upper limit is set by the kinetics of decomposition processes: as SOM accumulates, the production of CO_2 increases (assuming first-order kinetics), resulting eventually in the equilibration of C inputs and losses. The lower limit, presumably, is determined by the size of the inert SOM pool (Scharpenseel and Becker-Heidmann, 1994). Even with degradative soil management practices such as alternate fallow–wheat systems in western Canada, the SOM content eventually approaches a new steady state as the inert C occupies a progressively higher proportion of total organic matter.

The limits to SOM change can be extended in some instances by redistribution or transport of SOM among sites on the landscape. For example, severe erosion can conceivably result in virtually complete losses of SOM. Conversely, repeated additions of organic materials, such as animal manure, can increase SOM almost indefinitely (Sommerfeldt et al., 1988). In both cases, however, the gains or losses of SOM occur through lateral exchange of C between sites.

Given that there are limits to SOM change, it follows that gains (or losses) can continue only for finite periods of time. This hypothesis has implications for the use of soils as a sink for atmospheric C. For example, reduced tillage intensity and improved crop sequences have been proposed as means of reducing atmospheric C (Kern and Johnson, 1993; Lee et al., 1993; Varvel, 1994). Although such changes in management practices can promote SOM accumulation under many conditions, increases in organic matter may be of short duration (e.g., several years or decades) and cannot be long-term solutions to CO_2 accumulations from fossil fuel combustion (Schlesinger, 1990; Sauerbeck, 1993; Cole et al., 1993). Only during the transition between steady states do soils act as C sinks (Fig. 12.2), and once a new steady state is attained, no further gain is possible without some additional disruption.

IV. EFFECT OF SOIL ORGANIC MATTER CHANGE ON SOIL QUALITY

The quality of a soil can be evaluated only with respect to specific functions (Blum, 1994). In agriculture, soil quality is often defined in terms of its productivity (i.e., capacity to support crop growth). But soils also serve other essential functions, such as filtering hazardous compounds (Doran and Parkin, 1994), that are equally valid as a basis for defining quality.

However soil quality is defined, whether on the basis of productivity or other functions, researchers usually imply a positive linear relationship between SOM and soil quality. Thus, if a new practice increases SOM content, we conclude that it enhances soil quality; if a cropping regime results in SOM depletion, we infer a decline in soil quality. The same rationale is often applied to changes in SOM composition; an increase in labile SOM fractions (e.g., mineralizable N, light fraction C, microbial C) is interpreted as an improvement in soil quality. These relationships, however, probably apply only within certain limits. For example, beyond some threshold SOM content, further increases may no longer improve productivity (Janzen et al., 1992b). At some point, increases in SOM may even be considered detrimental to soil quality, particularly when considering soil functions other than crop production. For example, high SOM content in soil amended with excessive manure rates may be associated with high nitrogen concentrations, leading to nitrate leaching and reduced quality of that soil as an environmental buffer (Chang and Janzen, 1996). Whether a change in SOM affects soil quality therefore depends on the current level relative to some optimum; an increase in SOM will improve soil quality only if current levels are sub-optimal.

Each of the various functions ascribed to a soil may have a different optimum SOM content. For example, the optimum SOM content for maximizing crop production may be different from that for stabilizing pesticides or for scrubbing excess atmospheric CO_2. The optimum SOM content for a given soil will therefore depend on the suite of functions considered and the relative value assigned to each.

Even if we consider only one function, the optimum SOM content will still vary widely. For example, from the sole perspective of maximizing productivity, the optimum SOM content for a given agroecosystem may be influenced by the following factors:

1) *Composition of SOM*: The composition of SOM may be more important than its total concentration for promoting crop growth. Relative to inert SOM, for example, labile SOM has a disproportionate effect on nutrient-supplying capacity. It

may be, therefore, that a soil with high total SOM content, most of it inert, may be less productive than a soil with comparatively low SOM, much of it decomposable.

2) *Potential net primary production (NPP) or yield*: In general, a higher NPP will place greater demands on SOM and therefore result in a higher optimum SOM content. In parallel sites in Saskatchewan, for example, more-favorable moisture conditions result in higher production at Melfort than at Swift Current (Table 12.1). Consequently, the optimum SOM content is presumably higher at the former site. Differences in production among cropping systems also imply that the optimum SOM content varies with cropping practices, even at the same site. For example, because primary production is lower in a fallow–wheat system than in continuous wheat (Table 12.2), the former places less demand on SOM, and its optimum value may be lower. Consequently, the lower SOM content usually observed in a fallow–wheat soil does not necessarily imply a loss of soil quality, at least from the standpoint of productivity in fallow–wheat.

TABLE 12.1

Comparison of climate, productivity, and soil organic matter at two sites in Saskatchewan (from Campbell et al., 1997 and Campbell and Zentner, 1997)

Site	MAT[z] (°C)	Moisture deficit (mm)[y]	Productivity[x] ($Mg\ ha^{-1}$)	Soil C[x] ($Mg\ ha^{-1}$)
Melfort	0.8	96	1.8	65.4
Swift Current	3.5	371	1.3	34.3

[z]Mean annual temperature.
[y]Potential evaporation–precipitation.
[x]Productivity refers to long-term mean grain yield in a fertilized continuous wheat system. Soil C refers to organic C to a depth of 15 cm in the same treatment.

TABLE 12.2

Grain yield and soil organic matter characteristics from three long-term cropping systems at Swift Current, Saskatchewan (adapted from Campbell and Zentner, 1993 and Biederbeck et al., 1994)[z]

Cropping System	Grain[y] yield	Total soil C	LF-C	Min. N	Min. C
	$Mg\ ha^{-1}$	—— g C kg^{-1} soil ——		— mg N or C kg^{-1} soil —	
Fallow–wheat	0.94	17.2	1.2	80	158
Fallow–wheat–wheat	1.09	18.9	1.6	96	184
Continuous wheat	1.32	21.3	3.2	126	371

[z]All soil analyses are from 0–7.5 cm layer (from Biederbeck et al., 1994). LF-C = light fraction C, Min. N = mineralizable N (16-wk incubation); Min. C = mineralizable C (30-day incubation).
[y]Grain yield averaged over rotation phases (from Campbell and Zentner, 1993).

3) *Supplementary energy inputs*: The dependence of productivity on SOM can be reduced by the addition of supplemental energy. For example, a low nutrient-supplying capacity can be alleviated by applying fertilizer, and low moisture-holding capacity by irrigating, both of which require additional energy input. Consequently, the optimum of SOM content typically declines with increasing inputs of supplemental energy. In some very intensive agricultural systems, for example, there may be little relationship between SOM and soil productivity.

4) *Other soil and climatic conditions*: Optimum SOM contents may also be influenced by numerous other factors, such as soil texture, pH, moisture, and temperature. For example, if mineralization of nutrients is an important function, then optimum SOM levels may be much higher in cool soils, in which mineralization is constrained by temperature, than in warmer soils, which permit rapid turnover of the existing SOM.

Even when considering only one function (productivity), as in the preceding example, identifying optimum SOM values is exceedingly complex. If we add to this the need to consider other equally important soil functions (e.g., role as environmental buffer), then the objective of identifying optimum values becomes wholly unrealistic. More appropriate may be an effort to understand the demands placed on soil in specific ecosystems, and then to determine what changes in SOM content and composition would enhance the capacity of that soil to fulfill those functions.

V. CONCLUSIONS

Agricultural practices have exerted a profound influence on the dynamics of SOM. The most drastic effect may have been the initial introduction of arable agriculture, but changes in SOM are now usually influenced more by current management practices rather than by residual effects of initial cultivation. Adoption of revised practices offers potential for both gain or loss of SOM. Whether or not such changes affect soil quality depends on the functions ascribed to the soil and the demands the particular agroecosystem places on SOM. Within limits, practices that elevate SOM content will enhance soil quality by promoting productivity, reducing reliance on supplementary energy, and minimizing leakage of pollutants into adjacent environments.

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Chapter 13

SOCIOECONOMICS IN SOIL-CONSERVING AGRICULTURAL SYSTEMS: IMPLICATIONS FOR SOIL QUALITY

M. BOEHM and S. BURTON

I. INTRODUCTION

Farmers produce plant and animal crops in complex farming systems that must be, in the long term, environmentally, economically, and socially sustainable. Farming systems are dynamic and evolve as a function of the environmental, historical, social, economic, and political conditions of their time and place. Farmers must weigh the relative importance of environmental sustainability, which includes the maintenance of soil quality, against economic and social factors that also determine the short- and long-term sustainability of their enterprise.

Soil scientists involved with soil quality in agricultural systems have been concerned primarily with the environmental sustainability of agriculture. They have viewed soil conservation as mainly an environmental problem and have sought technical solutions to the problems of soil degradation. However, there has been poor adoption of many conservation practices and a continued degradation of agricultural soils despite the allocation of considerable resources, both nationally and internationally, to research and the development of soil-conserving practices. The concept of sustainable development, as outlined in publications such as *Our Common Future* (World Commission on Environment and Development, 1987), reminds researchers that the social and economic effects of human activities were no less crucial to their sustainability than the environmental effects. Researchers now recognize that soil conservation is as much a social and economic problem as it is a technological problem (Lovejoy and Napier, 1986; Gameda and Dumanski, 1995), and that practices that are not economically beneficial or do not conform to social or cultural requirements of the farm community cannot be adopted by farmers (Smyth and Dumanski, 1995). Dumanski and Smyth (1993) suggest that sustainable agriculture requires that socioeconomic concerns and environmental principles be integrated into new policies, technologies, and activities designed to improve agricultural systems.

Understanding the role of people and society in the adoption of conservation strategies is more difficult than providing the technological solutions (Swader, 1994). The important question now is how to develop conditions that convert the idea of wise management of soil resources into practices for farmers. This is a challenge, because programs to regulate and enforce the use of conservation practices will be ineffective if the differences among individuals within a culture and between cultures are not recognized (Swader, 1994).

Rural sociologists and socioeconomists have always regarded conservation as more than a technical problem. Soil degradation, however, has not traditionally been an issue in socioeconomic research. Since most farmland is privately owned, it was assumed that soil degradation leading to reduced crop productivity would be detrimental to the interests of farmers. Conventional wisdom held that education would result in voluntary adoption of soil-conserving practices (Buttel et al., 1990). The diffusion adoption approach, used originally in the 1950s and 1960s to study the adoption of commercial technologies by farmers, was also used to study the adoption of conservation technologies. It was assumed that farmers who were educated about soil-conserving technologies, had access to agronomic and economic information about them, and received extension assistance in their adaptation to local climate, soil, and social conditions would adopt the prescribed practices (Nowak, 1984; Heffernan and Green, 1986). Research has shown, however, that the diffusion adoption model does not apply to adoption of non-commercial technologies, and that education is not sufficient to ensure adoption of conservation practices that are not profitable (Pampel and van Es, 1977; Buttel et al., 1990).

Most farmers do not lack information about the benefits of soil-conserving practices (Clay and Lewis, 1990; Napier et al., 1994), but they may be unable or unwilling to change their farming practices if the proposed conservation techniques improve soil quality but not economic and social sustainability (Lovejoy and Napier, 1986; Bradsen, 1994).

Profitability is one of the most important factors governing the adoption of soil-conserving practices. If the costs of conservation practices exceed the short-term, and possibly, the long-term benefits, farmers have no incentive to adopt them, whereas economically profitable technologies with complementary environment-enhancing characteristics are readily adopted (Camboni and Napier, 1994; Cary, 1994). The vagaries of weather, biology, and volatile markets mean that a significant number of farmers will always be in financial stress, averse to taking risks (Ehrensaft and Bollman, 1986), and reluctant to implement changes that may reduce yield and net income or increase capital costs (Hexam et al., 1979; Reicosky et al., 1995). Willingness to undertake risks is related positively to a farmer's income and wealth position, which have been deteriorating in many parts of the world since the 1980s (Lo and Sene, 1989; Stonehouse, 1994).

Increased grain production worldwide and decreased trade, combined with a trade war between the U. S. and the European Economic Community from the 1970s to the early 1990s, have altered the structure of world agriculture (Fulton et al., 1989). As a result, farm incomes have declined over time, and many farmers, especially subsistence farmers, do not have the economic resources to implement soil conservation programs (Camboni and Napier, 1994). Poor people have the clear objective of maximizing income, usually with limited means (Seckler, 1986). For example, Southgate (1994) suggests that in the Ecuadorian Andes, where farmers with small land holdings have faced declining commodity prices and profits for 20 years, soil degradation is a form of depreciation of assets. For farmers with no access to credit, it is rational to allow fixed assets to depreciate before finding new work. In Rwanda, Clay and Lewis (1990) found that farmers faced with limited land and capital and low commodity prices were compelled to maximize production within each growing season at the expense of long-range environmental stability. In many developing countries, acute social and economic problems requiring short-term solutions foster a certain degree of indifference towards long-term environmental conservation in economic planning efforts (Bandara, 1989).

In this highly competitive world market, farmers in industrialized economies who have access to credit view the purchase of technology to increase production or decrease costs as an efficient and rational production decision. The result has been increased mechanization and specialization, continuous production in excess of demand, and larger and fewer farms (Gertler, 1992; Vassey, 1992; Walker and Young, 1986, Pimentel et al., 1976). As agriculture has become more capital-intensive, there has been an increase in purchased inputs, such as fertilizer and pesticides (Vassey, 1992). Increased use of inputs has made crop yields larger and more uniform and has changed the economic structure of the entire agricultural sector (Tamm, 1991). Farmers in capital-intensive systems often operate with a high proportion of borrowed capital and are required to pay interest costs on a timely basis (Stickel, 1990; Gilpin et al., 1992). Borrowed capital, coupled with overproduction and the cost–price squeeze, has caused farmers to plan for the short term while heavily discounting the long-term gains from soil conservation (Van Kooten and Furtan, 1987). Most of these farmers make management decisions at the economic margin and will not adopt conservation practices that reduce their ability to produce a profit to pay for equipment and inputs (Stickel, 1990).

In addition to profitability, adaptation to local cultural and social conditions is an important characteristic of readily adopted farming practices. Soil-conserving practices that are technically feasible and adaptations or improvements of existing practices tend to be well received by farmers. In the West African Sahel, in the country of Burkina-Faso, construction of micro-catchments for water harvesting in farmers' fields was successful at improving yields and reducing soil erosion and was readily adopted when the catchments were built of stone (Reij, 1994). Stone bunds were used traditionally in the area, but their efficiency was limited, because they were not constructed on the contour and gaps were left between the stones. The traditional practices were improved using a water tube to determine contour lines and better construction techniques, which farmers easily mastered. Reij (ibid.) reported that tens of thousands of hectares have now been treated with stone bunds, and their construction is limited only by the availability of stones.

Seckler (1986) describes a project in Punjab, India, that was designed to control soil erosion and provide irrigation water by construction of a check-dam to control flooding. The project succeeded because the allocation of water rights was done according to local social custom. Each household in the village was allocated an equal share of water. Landless households, which grazed animals on communally owned land, were free to barter or sell their share in exchange for adhering to prescribed grazing practices within the watershed. Soil erosion decreased and the irrigation system increased the agricultural income of the area.

In North America, soil-conserving practices that are compatible with the trend towards large, mechanized farms, such as minimum or zero tillage, have been widely adopted. Minimum- and zero-tillage practices are often adopted by farmers because they reduce energy use and the cost of production rather than because they conserve soils (Anderson and Bray, 1995; Buttel et al., 1990). Other recommended practices, such as inclusion of legumes in rotations, cover crops, shelter belts, use of animal manures and grassed runways, which restrict the use of large equipment, have not been widely adopted by farmers with large land holdings (ibid).

The following case studies were carried out to identify links between socio-economic parameters, use of soil-conserving farming practices, and soil quality characteristics. The studies, done in Nepal and western Canada, identified some socioeconomic characteristics of farmers who used soil-conserving practices, as well as some of the social and economic factors that favoured continued use of more traditional practices, despite their negative effects on soil quality.

II. CASE STUDIES

Rotations with legumes or trees in Chitawan, Nepal, and systems with reduced tillage and long rotations in Saskatchewan, Canada, can improve soil quality compared to the more traditional farming practices of each region. Case studies of the adoption of these soil-conserving systems show that many of the economic and social factors that determine adoption were similar in both regions, despite their vastly different cultural and biophysical environments. Both in subtropical Chitawan, where subsistence agriculture is land- and labour-intensive and dependent on resources within the farm or community, and in temperate Saskatchewan, where farming is commercial, mechanized, extensive, and dependent on marketplace interactions, farmers' decisions about farming practices are based on the availability of land, labour, and capital, and their willingness to increase risk or potentially reduce production.

A. Case study 1: Land use management options in Chitawan, Nepal

In Nepal, subsistence farming is dependent on a delicate ratio of people, livestock, and forest. Agricultural land is used very intensively, with widespread inter-cropping and double or triple annual crop rotations. Is such intensive use causing soil deterioration and is this high crop production sustainable, especially when inputs are often limited? Is soil conservation a priority when the main concerns are providing food for the family and farm animals, minimizing risk, and obtaining the land, labour, and capital needed to produce a crop?

1. Study area

The study was located in the Pithuwa area of Chitawan, central Nepal. The Chitawan is a *dun* (tectonic valley) in the Siwaliks ranges. The soils (young phases of Eutrocrepts and Hapldolls) were developed on alluvial fans and aprons and have a high capability for both agriculture and forestry (L.R.M.P., 1986). The climate is humid and subtropical, with a hot, rainy monsoon season; a warm, dry winter season; and a hot, dry, windy pre-monsoon season.

In 1985, Pithuwa covered 1,200 ha and supported 7,100 people or 1,200 households. The area has a mixture of castes and religions, with people resettled from the Middle Mountains; most people speak Nepali and depend on agriculture for their livelihood. The farms studied were representative of the land tenure pattern, population, and land use.

2. Methods

Informal methods were used to interview 25 key informants (Chambers, 1985; Rhoades, 1985), followed by 75 farm interviews for detailed assessment of land use and farm management, including family characteristics, land holdings, food sufficiency, fuel wood use, livestock, land use, and crop production. An adapted enterprise budgeting method (Harsh et al., 1981; Kay, 1986; Burton, 1987) was used to evaluate land use options in economic terms, using an interactive PC-Lotus Symphony program. The relative profitability or gross margins for eight major crops were compiled into eight common cropping combinations to estimate gross margin by crop rotation. This cropping system model was first run using data for average conditions, yields, and prices. Scenarios were then applied to the crop budgeting model to test the sensitivity of crop income to selected factors of production. Environmental conservation and decision-making approaches were used to integrate soil quality and farming systems and to develop recommendations for land use management (Burton et al., 1990).

3. Farm system characteristics

Families were relatively large, reflecting the invaluable labour contribution of family members. Livestock, used for draft power, were present in high numbers, but their health was poor. Interviewed farmers described 13 crop rotations and referred to their land use types in two categories: *khet* or puddled, irrigated lowland used for rice-based cropping rotations; and *pakho* or well-drained, rain-fed upland used for maize-based rotations. Four irrigated rice-based systems and four rain-fed maize-based systems were widespread in Pithuwa (Fig. 13.1). On irrigated lowland, late rice was traditionally followed by a winter crop (late rice–mustard or late rice–lentils). Early rice could be followed by wheat or mustard and a third crop of early spring maize (early rice–wheat–maize or early rice–mustard–maize). On rain-fed upland, late maize was traditionally followed by mustard (late maize–mustard). Rotations with three crops per year, such as maize followed by potato relay cropped with wheat

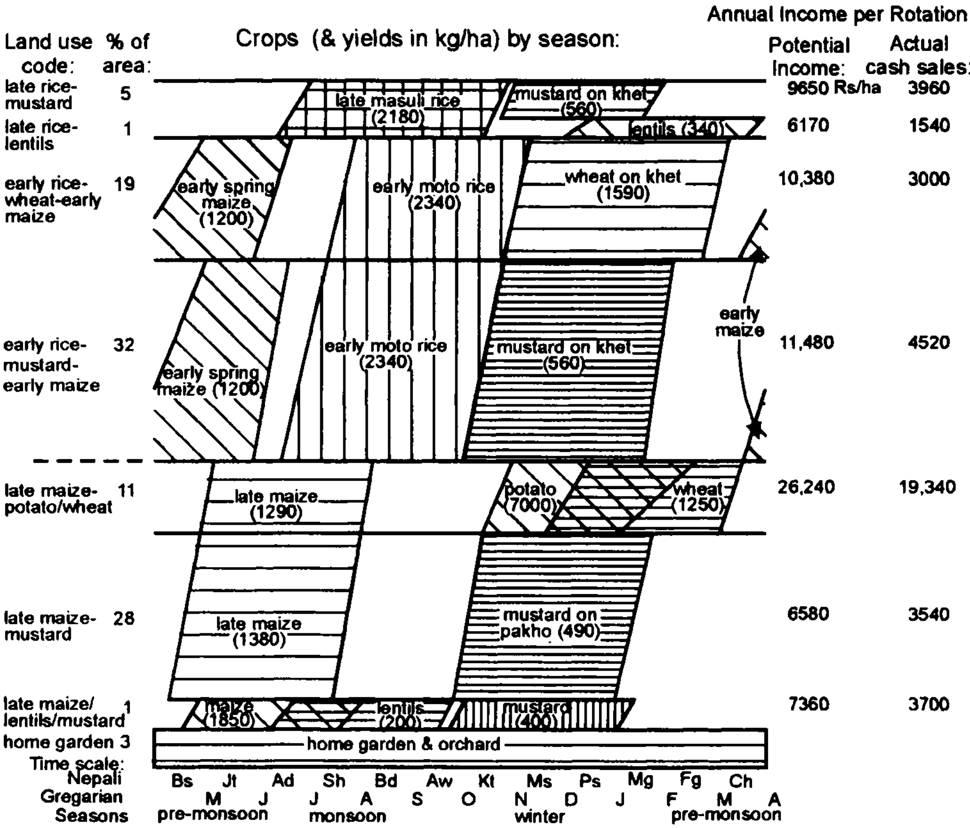

Fig. 13.1. Predominant cropping rotations, crop yields, potential income, and actual cash sales in the Pithuwa area of Chitiwan, central Nepal, 1985.

(late maize–potato/wheat), were also common. Although not widespread within the study area, cropping systems with legumes (late rice–lentils or maize/lentils–mustard) and an agroforestry system from a nearby area were included because of their soil-conserving nature.

In this area, rice and maize are staples in the family diet. Residues of rice, maize, and wheat are staples for farm livestock. Lentils are an important protein supplement to human and animal diets. Income from actual cash sales of these four crops is small, because they are highly valued on the farm. In contrast, mustard and potatoes are cash crops that are readily sold in the market, resulting in higher actual cash sales than rice, maize, wheat, or lentils (Fig. 13.1).

4. Productivity and profitability of crop rotations

Enterprise budgeting showed that overall profit levels of the farms were extremely low, all cropping systems were labour intensive, and operating capital requirements were low, except for the rotations that included potatoes (Fig. 13.2). The double crop

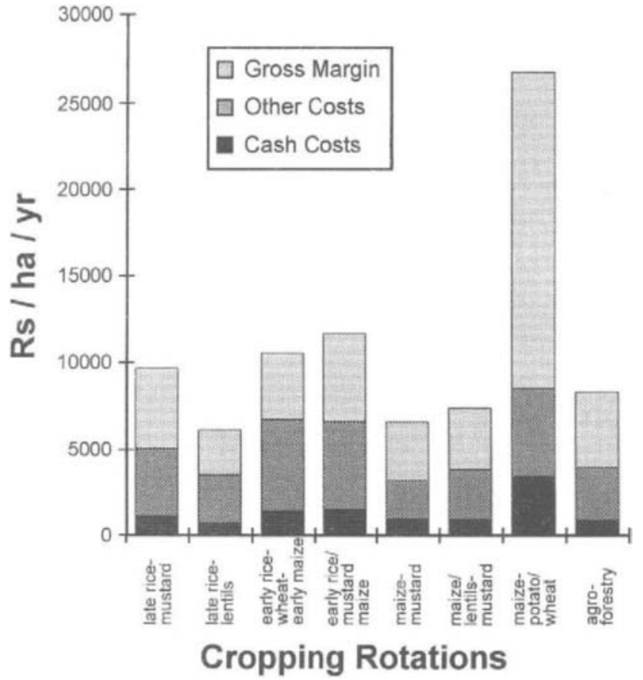

Fig. 13.2. Gross margins and costs of eight cropping rotations in the Pithuwa area of Chitawin, central Nepal.

rotation with late maturing rice followed by mustard was more profitable than the triple crop rotation with a new high-yielding *moto* rice variety followed by wheat and maize. However, in the monsoon maize rotations, triple crop rotations (maize/lentils–mustard, maize–potato/wheat) were more profitable than double crop rotations (maize–mustard). The three most profitable options (maize–potato/wheat, early rice–mustard–maize, late rice–mustard) were centred around the most profitable crops: potatoes, mustard, and late rice. The agroforestry system, which could supply all family fuel wood needs, was more profitable than similar rain-fed rotations without trees.

Adoption of high-profit crop rotations requires the appropriate land, labour, power, and capital resources. Rice-based crop systems require irrigated, bunded, puddled land, whereas rain-fed crop rotations require well drained, aerated soils for the monsoon maize crop. The agroforestry system requires high pH irrigation water to establish the trees. Labour and power requirements vary with cropping rotation and may constrain the adoption of innovative crop rotations. Triple cropping requires the greatest amount of labour (77 to 85 man-days year^{-1} and 165 to 175 woman-days year^{-1}) and power (48 to 60 bullock-days year^{-1}).

The large amount of operating capital required for the potato and triple rice rotations limited their adoption by poor farmers. For example, the highly profitable rotation with potatoes required 3,174 rupees for the purchase of seed and fertilizers.

In 1985, this represented about \$170 U.S. or double the mean annual Nepali family income. The rice-based triple rotations required considerable operating capital requirements as well. Farmers with low cash flow and no access to credit told us they were forced to stay with traditional double crop rotations, such as late rice followed by mustard or lentils and maize followed by mustard (Burton, 1987).

Several scenarios, including changes in yields, crop prices, input prices, or labour costs, were used to test the sensitivity of the farm profit or gross margin to variations in factors of crop production. Increased input prices were reflected in similar decreases in gross margin per crop rotation. In contrast, factors such as crop prices and crop yields had a two to three times greater effect on gross margin. The results of the sensitivity analysis support the opinion of several international researchers, who claim that the complexity of the farming system needs to be understood before introducing innovations to farmers that are operating at such narrow profit margins (Zandstra et al., 1981; Van der Veen, 1982; Schroeder, 1985). Direct effects, such as fertilizer prices or poor crop yields, become critical to the already complex double and triple cropping systems. There is also an indirect effect on cropping options by factors such as adequacy and quality of animal feed or the health of the farm family labour pool. This study reveals the complexity, diversity, and delicate balance of these farming systems.

5. Decision-making approach

The decision-making approach asks the following questions: "Who are the decision makers? How are land use management alternatives evaluated? Should conservation considerations be modified after understanding the farm decision-making process?"

Scientists can elucidate the importance of conservation processes and measures, and governments at all levels can legislate and enforce conservation guidelines, but decisions concerning land use management and conservation are made by the farmer in the context of family concerns and priorities. The farmer's main goals are to increase food production and ensure family health. The key informant interviews and the soil quality study suggest that six criteria are important for decisions about land management: crop preference, crop productivity, profitability, resource requirements (land, labour, draught power, and operating capital), soil quality indicators (pH, organic C, and available P), and relative risk (Table 13.1).

Choices based entirely on crop preference are straight forward. Rice and maize are the preferred staples for the family and livestock diets; therefore, the monsoon season is reserved for those two crops. Land uses with a third crop, either pre-monsoon or a relay crop, are often chosen to provide dietary diversity. Options that include a cash crop, such as mustard or potatoes, are also preferred.

In terms of crop productivity, rice is a high priority, but the highest yielding crop is potatoes. Based on this criterion alone, maize and wheat are the third and fourth crop choice. Thus, combinations such as maize–potatoes inter-cropped with wheat or early rice–wheat–maize are chosen above other lower yielding rotations such as maize–mustard and maize–lentils. When only profitability is considered, the rotation with potatoes is the first choice (annual gross margin 18,000 Rs ha^{-1}), followed by

TABLE 13.1

Rating of agricultural land use options, in Chitawa, central Nepal, using a ranking system[1]

Decision-making criteria	*Khet* land use codes[2]				*Pakho* land use codes[3]			
	kc	kd	kg	kh	pc	pd	pa	pi
Crop preference[4]	6	5.5	6	5.3	4	4.3	4.3	3.3
Productivity	4	3	6	5	1	2	8	7
Gross margin (relative profit)	6	1	4	7	2	3	8	5
Resource requirements[5]	3	6.8	2.5	2.8	6.7	5.7	1.8	4.9
Soil quality (organic carbon)	2	1	6.5	6.5	3	4	5	8
Risk rating	7	7	4.5	2	7	4.5	2	2
Overall mean rank	4.7	4	4.9	4.8	3.9	3.9	4.8	5
Rank of mean ranks	4	3	7	5	2	1	6	8

[1]Rank of 8 = best or most desirable land use option; rank of 1 = worst or least desirable land use option.
[2]kc = late rice–mustard; kd = late rice–lentils; kg = early rice–wheat–early maize; kh = early rice–mustard–early maize.
[3]pc = late maize–mustard; pd = late maize/lentils–mustard; pa = late maize–potatoes/wheat; pi = agroforestry.
[4]Mean rank of crops of all seasons.
[5]Mean rank of land, labour, power, capital.

the irrigated triple crop rotations early rice–wheat–maize or early rice–mustard–maize. The choice is more complex when the actual amount of crop sold on the market is considered. More mustard and potatoes are sold for relatively higher prices.

Rotation choices varied with available resources (land, labour, power, and capital). Irrigated land is required to grow the more profitable triple rice-based rotations, whereas the late maize-based rotations require well-drained, rain-fed land. If labour is limited, the traditional maize–mustard rotations (110 days ha^{-1} yr^{-1}) would be chosen over the labour-intensive triple- cropping systems (235 to 245 days ha^{-1} yr^{-1}. Men and women had defined roles within the agricultural labour pool, and women often worked more days in the fields than men (one and a half times more for traditional rotations to two times more for innovative rotations). Rotations that conserve soil will not be adopted if the male to female balance of labour required is different from the available gender mix.

Operating capital includes cash requirements for fertilizers and taxes, but not bullock power, compost, or seed. If cash is limited, traditional land uses such as late rice–lentils or maize–mustard are better choices than innovative triple rotations (Fig. 13.2).

Based on the soil quality indicators (soil pH, organic C, and available P) and more detailed soil analysis (Burton et al., 1989), the more innovative land uses were relatively more soil conserving. Unexpectedly, soil was conserved rather than degraded under the intensive triple-cropping practices, for two reasons. First, these rotations included a pre-monsoon crop during the critical season for erosion control. Secondly, they produced higher yields, which justified higher additions of fertilizer and residues to the land. In contrast, the expected improvement in soil properties with the inclusion of legumes did not occur. Nitrogen fixation may have been inhibited by drought. Companion crops or farm family utilization may have reaped the nitrogen or protein increases rather than the soil.

Risk ratings were based on the variability of crop yields, the sensitivity of a particular crop to drought, infestation, and disease, and market price fluctuations (Burton, 1987). The more traditional land uses (late rice–mustard, late rice–lentils, maize–mustard) involved the least risk, whereas the more innovative rotations (early rice–mustard–maize, maize–potato/wheat, agroforestry) were riskier. Based on the farm interviews, it seems that farm families close to subsistence level chose options with the least risk.

In the actual farm decision-making process, all six factors were evaluated simultaneously. The differences were rather small, but the benefits of agroforestry and triple-crop rotations outweigh their disadvantages if all of the criteria are weighted equally (Table 13.1). Different families had different priorities and selectively weighted the criteria. For example, small farms would focus on crop preference or risk; and they often had less access to bullock power, cash, or better quality *khet* land. Their management decisions differed from larger farmers, who focussed on crop production, profitability and soil quality.

Although an oversimplification, this method provides a flexible tool to understand the land use decision-making process. The final recommendations for land-use management have to focus on the trade-offs. Land use options that promote soil conservation usually require more labour, power, irrigation, and operating capital. There are also higher risks associated with these soil-conserving land uses because of price and yield fluctuations, and the preferred crops are often not included. Perhaps the question should be asked, what family goals and priorities are compatible with soil conservation recommendations? Soil-conserving land use alternatives (agroforestry, early rice–mustard–maize, early rice–wheat–maize, maize–potato/wheat) are usually more productive and profitable and the farmers with large land holdings could more easily accept them because they often have access to the necessary inputs. For many of the farmers with small land holdings, the disadvantages of risk and higher input requirements outweigh the benefits associated with growing such crops.

Given this decision-making context, we need to ask how the introduction of agroforestry would be perceived by the farm family. Agroforestry requires more woman-days of composting and tending the seedlings, but fewer woman-days of fodder and fuel collection from the distant forest. In contrast, more man-days are required, with considerable initiative to attend workshops in another district and visit the nearest nursery for seedlings (several hours' walk from Pithuwa).The multipurpose trees have many side benefits throughout the whole farming system, so with promotion, such as sponsoring workshops, the constraints can be overcome. Extension messages to promote soil conservation and agroforestry need to influence women's practices (spreading compost or planting trees), as well as men's practices (buying fertilizer or tree seedlings).

B. Case study 2: Adoption of continuous cropping practices in Saskatchewan, Canada

Prairie soils have degraded since their conversion to agriculture. Moisture deficit during the growing season in the Canadian Prairies created a dependence on the traditional rotation of drought-tolerant spring wheat followed by summerfallow. Tillage summerfallow, used to control weed growth and store soil moisture, thereby reducing the risk of crop failure due to drought (Campbell et al., 1990; Lee and Pankhurst, 1992), is the most harmful farming practice on the Prairies (Larney et al., 1994). Soil scientists have identified minimum tillage and continuous cropping practices that can maintain higher soil quality than the crop–fallow system, but these practices have not been adopted by all farmers (Janzen, 1987; McKenzie et al., 1992; Lal et al., 1994). In the 1991 agricultural census, 29% and 14% of Saskatchewan farmers reported using conservation tillage and zero-tillage systems, respectively (Statistics Canada, 1991).

1. Methods

The study involved 16 farms located about 40 km west of Saskatoon, Saskatchewan, Canada, in a semi-arid area with Dark Brown Chernozem (Haploboroll) soils developed on sandy-loam glaciolacustrine deposits. The farming systems studied were a two-year crop–fallow rotation, an extended rotation of four to five crops followed by summerfallow, and continuous cropping for at least 10 years (Boehm and Anderson, 1997). A case study approach based on farmer interviews and a questionnaire was used to determine the characteristics and attitudes of the farmers associated with each of the farming systems, as well as the characteristics of each of the farm types. Agricultural census data, obtained from Statistics Canada for 1971 to 1991, was cross-tabulated by the amount of summerfallow used on each farm. Census data were provided for the rural municipalities (RMs) where the case study farmers lived (RMs 345 and 346), excluding farms with operators over 70 years of age, farms smaller than 128 ha (320 acres), and farms with no land used to produce annual crops (Boehm and Anderson, 1997).

2. Farm and farmer characteristics

Since 1971, the proportion of farmers using a continuous cropping system rather than a crop–fallow system has increased (Table 13.2). Compared to crop–fallow farms, the continuously cropped farms tended to be more diversified than the crop–fallow farms, with proportionally less area seeded to wheat (Table 13.2), more cattle (Table 13.3), and relatively less land used for annual grain production.

A majority of farmers in all of the groups used chemical inputs, but fertilizer use was lowest on crop–fallow farms (Table 13.2). This may be because older farmers were more likely to use a higher frequency of fallow (Table 13.3), which is traditionally a low-input system. Among the case study farmers, the crop–fallow system was used by older farmers who did not want to change systems shortly before retiring, and by beginning farmers who had limited resources and wanted to use a system that reduces risk of crop failure and requires the least management skills and inputs. The crop–fallow farmers also had higher rates of off-farm employment

TABLE 13.2

Characteristics of each summerfallow (%SF) group since 1971, in a semi-arid region of Saskatchewan, Canada (Boehm, 1996)

%SF Group	Rms 345 and 346[1]			
	1971	1981	1986	1991
	Farms (% of total)			
< 10 Continuous crop	4	9	11	21
10–24 Extended rotation	8	8	18	16
25–45 Extended rotation	56	46	50	45
> 45 Crop–fallow	33	37	21	19
	Farm land in field crops (%)			
< 10 Continuous crop	76	66	71	72
10–24 Extended rotation	64	68	68	70
25–45 Extended rotation	54	52	56	56
> 45 Crop–fallow	44	45	46	47
	Wheat as a % of field crops			
< 10 Continuous crop	41	48	52	46
10–24 Extended rotation	33	53	57	53
25–45 Extended rotation	59	59	61	59
> 45 Crop–fallow	41	74	74	71
	Farms that use fertilizers (%)			
< 10 Continuous crop	18	46	61	68
10–24 Extended rotation	6	67	73	77
25–45 Extended rotation	21	57	77	66
> 45 Crop–fallow	22	62	68	56
	Farms that use herbicides (%)			
< 10 Continuous crop	23	46	69	72
10–24 Extended rotation	43	71	88	84
25–45 Extended rotation	51	78	89	76
> 45 Crop–fallow	54	75	87	77

[1]Rms refers to Rural Municipalities.

(Table 13.3), indicating that the choice of a crop–fallow rotation may be based partly on a lack of available labour for farm work.

The continuous crop and extended rotation farms in the case study were larger than farms with a comparable proportion of summerfallow in the 1991 census (Table 13.3), which suggests that the case study farms may not have been representative of the average long-rotation farm in RMs 345 and 346. The continuous crop farmers selected for the case studies had fields that had been cropped continuously for 10 to 20 years, so they were early adopters of this practice and are probably more innovative than the average farmer who began continuous cropping more recently.

A comparatively large proportion of the continuous crop farms were managed by more than one family, such as parents and an adult son, or brothers and their

TABLE 13.3

Characteristics of each summerfallow (%SF) group in Rms 345 and 346 from 1991 census data and the 1993 sample in a semi-arid region of Saskatchewan, Canada

Characteristics	SF%	Group	Census data 1991	Case-study farms 1993[1]
Age (yrs)[2]	10	Continuous crop	43	
	24	Extended rotation	45	
	45	Short rotation	48	
	>45	Crop–fallow	49	
Off-Farm Work (% of Farmers)	10	Continuous crop	40	20
	24	Extended rotation	39	50
	45	Short rotation	34	
	>45	Crop–fallow	42	40
Farm size (ha)	10	Continuous crop	436	1290
	24	Extended rotation	749	943
	45	Short rotation	503	
	>45	Crop–fallow	374	532
Rented land (%)	10	Continuous crop	35	33
	24	Extended rotation	40	30
	45	Short rotation	39	
	>45	Crop–fallow	42	44
Farms with cattle (%)	10	Continuous crop	53	66
	24	Extended rotation	44	40
	45	Short rotation	36	
	>45	Crop–fallow	31	0

[1]The case study did not include any short-rotation farms.
[2]Age was not reported for the case study farms, because several of them comprised more than one generation, making averaged data misleading.

families. On a per-family basis, the continuous crop farms averaged about 800 ha per family, which is close to the mean farm size for the area. The proportion of rented land was least for the continuous crop farms and highest for the crop–fallow farms in both the census data and the case-study (Table 13.3).

Crop expenses and total farm business-operating expenses were highest for continuous-crop farms and lowest for the crop–fallow farms, based either on total farm area or seeded area (Table 13.4), mainly because of greater inputs of fertilizers and herbicides, per hectare, in the continuous-crop system. In 1986, sales minus expenses tended to increase as the amount of summerfallow increased, but in 1991 this trend was reversed (Table 13.4). Possible reasons for the increased profitability of continuous crop farms from 1986 to 1991 are increased yields in 1991 because of greater experience in crop production using a continuous-crop system; improvements in equipment designed for seeding into stubble; and improved soil quality over time, especially increased water-storage capacity, better nutrient cycling, and

TABLE 13.4

Expenses data from the agricultural census for Rms 345 and 346 for 1986 and 1991, based on total land area and area seeded to annual crops for each of the % summerfallow (%SF) groups in a semi-arid region of Saskatchewan

%SF	Group	Total farm basis		Annual crop-area basis	
		1986	1991	1986	1991
		Crop expenses ($ ha^{-1})[1]			
< 10	Continuous crop	33	27	47	37
10–24	Extended rotation	32	30	47	43
25–45	Extended rotation	24	20	43	36
> 45	Crop–fallow	16	15	35	31
		Total expenses ($ ha^{-1})[2]			
< 10	Continuous crop	225	313	317	432
10–24	Extended rotation	155	149	227	214
25–45	Extended rotation	119	116	213	205
> 45	Crop–fallow	85	101	185	216
		Sales–expenses ($ ha^{-1})[3]			
< 10	Continuous crop	25	71	34	98
10–24	Extended rotation	31	31	45	41
25–45	Extended rotation	45	33	80	58
> 45	Crop–fallow	44	17	95	37

[1]Fertilizer, herbicides, insecticides, fungicides, seed, seed treatment, and seed-cleaning costs.
[2]Crop, rent, wages and salaries, interest, machinery and fuel, livestock, custom and contract work, telephone, and electricity expenses.
[3]Sales from agricultural products, marketing board payments, program and rebate payments, dividends from co-operatives and custom work minus the operating expenses.

increased microbial activity. Relatively greater price increases for cattle compared to wheat from 1986 to 1991 would also increase income on farms with cattle.

3. Factors that influence farmer attitude

One of the major factors that influenced farming practices was the availability of labour and financial resources on each farm. Farmers with financial or labour shortages were limited in their ability to adopt new practices that require a capital investment and were less willing to undertake risk. Financial and labour limitation were often a function of the farmer's age. Farmers nearing retirement age were limited more by lack of education and labour than by financial resources. Often, they were not aware of the negative effects of summerfallow and excessive tillage on soil properties, even on sandy soils that are susceptible to wind erosion. Young farmers who had recently started farming were generally aware of the conservation and economic benefits of longer rotations, more diversified and higher-input farming systems, but were unable to adopt them because of lack of capital. A common strategy adopted by these families to increase cash flow was off-farm employment, which increased income but reduced the amount of time and labour available for the farm operation. Because of the shortage of labour, these farms usually produced only grain crops, generally in a crop–fallow system, which requires the least labour input at seeding and harvest time, the least chemical inputs, and the least risk.

Multi-family farming is a strategy that provided more opportunity for young farmers. Seven of the case study farms were multifamily farms (all of the continuous crop farms, two of the extended rotation farms and one crop–fallow farm), which had the advantage of more capital and labour resources than the single-family farms. Generally, the older generation provided experience and much of the capital in the form of cash, equipment, land, and buildings, while the younger generation supplied labour, along with information and skills gained through relatively recent post-secondary education. On the continuously cropped multi-family farms, individuals specialized in particular aspects of the operation, and as a result these farms were more diversified, with production ranging from mixed cattle and grain farms to seed and leaf-cutter bee production. The rate of off-farm employment was lower among the multi-family than the single-family farms, and the increased availability of labour and capital provided greater opportunity for diversification, experimentation, and change than on single family farms. Presumably, this is why the first farms to adopt continuous cropping systems in RMs 345 and 346 were all multifamily farms.

On the continuously cropped farms, a diversity of crops, both plant and animal, was produced to ensure adequate farm income, to minimize financial risk, and to use labour resources efficiently. Many of these farmers considered the production of livestock, in addition to cash crops, to be crucial to their success. They raised livestock in order to earn a sustainable return from poor quality lands, usually sandy-textured soils that are susceptible to erosion and not suited to the production of annual crops but are suitable for hay and forage production. These farmers attributed their success with longer rotations to cattle income, which they used to subsidize their grain operations if yields were poor in the early years of the transition from a fallow-based system. The continuous crop farmers were less concerned about the increased risk of relatively low yields in dry years, because they could rely on some income from cattle. The experienced continuous crop farmers indicated that crop yields tend to be more variable for the first five years because of their inexperience with the new system, in which timing, machinery needs, tillage requirements, and input requirements are all different than in the traditional crop–fallow system. One farmer suggested that during the first years of the transition, the benefits of summerfallow are lost, but the increased moisture storage and nutrient-cycling ability of the longer rotation soils have not yet been achieved.

Management skills varied among the farmers but were difficult to determine. Experience, personality, and aptitude vary among individuals in all professions, bestowing greater ability on some than on others. The farmers who managed continuous crop, mixed farms that produced a variety of cereal, pulse, oilseed, and animal crops, had the skills and desire to manage relatively complex and diverse farms, whereas other farmers wanted a simple system that required relatively little management, such as straight grain production in a cereal grain–fallow rotation with relatively low fertilizer inputs and the use of both herbicides and tillage to control weeds. The desire to pursue employment or recreational activities in addition to farming was often a major motivation for those farmers who used a traditional straight crop–fallow system.

All the farmers used economic arguments to justify their choice of farming system. Differences in the availability of resources determined which system would be most profitable for each of the farmers. Those who could afford the increased labour and financial costs of seeding all of their land, along with the increased risk of reduced yields per seeded land area in a continuous cropping system, felt they were better off obtaining a return from all of their land. Farmers who were limited in labour, money, or both, believed that their best option was to put labour and input costs into crops seeded on fallow land, where they were most likely to obtain maximum yields. Avoiding risk was an important factor, particularly for those with a large amount of borrowed capital. The economics of the different systems are thus dependent on the resources available to the farmer.

III. CONCLUSIONS

Agricultural systems reflect the climate, natural resources, availability of energy and inputs, and culture of their region. In the subtropical parts of Nepal, favourable climate makes two or three crops per year possible under irrigated conditions. Farming systems are dependent on human and livestock energy, and a family lives at a subsistence level from a few hectares. On the Canadian Prairies, cold winters and arid conditions limit crop production to one crop per year. Mechanization and high levels of capital make it possible to farm the hundreds of hectares required to support a farm family.

For both regions, farming systems and practices that can improve or maintain soil quality are understood, but they are not used by some farmers because of economic limitations or social factors. In both Nepal and western Canada, the most soil-conserving practices require higher levels of inputs, capital, and labour. Farmers who lack sufficient labour or access to capital continue to use the lower-input, lower-risk traditional farming systems of their area. Farmers who have the resources to adopt soil-conserving practices do so on the basis of their profitability and cultural compatibility. In Saskatchewan, continuous cropping is well suited to the high-input, highly mechanized systems already used by most farmers. In Nepal, farmers who manage larger areas of land will often adopt soil-conserving or innovative triple-crop options if they are more productive or profitable. However, other cultural factors, such as the change in gender distribution of labour with innovative cropping systems, or the preferred taste of the longer-maturing rice varieties, may affect the rate of adoption.

Successful soil conservation programs must include an analysis of the profitability, and land, labour, and capital resource requirements of the proposed conservation system. Production levels and potential risk are additional factors to consider for mixed or smaller farms in Saskatchewan or subsistence farms in Nepal. But most important, a flexible people-first approach that includes, as necessary, other management and decision-making criteria that affect the farm families existence is essential for effective soil conservation.

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Chapter 14

TOWARD A FRAMEWORK FOR SOIL QUALITY ASSESSMENT AND PREDICTION

K.J. GREER and J.J. SCHOENAU

I. INTRODUCTION

Determining the quality of a soil for a given use requires integration of many interrelated properties that control or mediate that use. Such a complex task requires a framework or strategy that not only categorizes the soil's fitness to function, but also provides a means to predict soil fitness under some other combination of soil attributes.

Previous attempts to describe soil quality have focussed on specific criteria or indicators that must exist in a "good" soil (Pettapiece, 1987; Mahnic and Toogood, 1992). All of these attempts have been impeded by: 1) the need for restriction by plant type and ecoregion, 2) disagreement among experts regarding which soil properties are fundamental, and 3) the inability to adequately express the interactions among the soil, plant, and ecoregion.

Recent approaches suggest that a group of intrinsic or critical health indicators exists for all soils, analogous to health indicators for humans, such as heart rate or blood pressure (Larson and Pierce, 1991). This simple approach appears to overcome many of the aforementioned problems. However, the human analogy does not apply to all soils. Humans have similar anatomy and function; therefore the criteria and standards indicating good human health vary little between individuals. Soils, on the other hand, support diverse plant communities under widely varying climates. A soil supporting halophytes has vastly different "health" criteria than a soil supporting glycophytes. Even within each of these plant groupings, the suitability of the soil differs. Thus, criteria for a good quality soil depend on the plant to be supported and the climate within which the soil functions. This is not to say that unifying factors do not exist; instead it is to make clear that a constant "critical level" does not exist for all soil–plant–ecoregion combinations.

Garlynd et al. (1994) attempted to build a correlation between farmers' observations of soil health and the soil properties controlling crop production. The goal of their "interpretive framework" was to relate production experience to specific soil data in an attempt to derive a minimum data set that would predict the soil fitness for crop production. Our chapter introduces another approach to assessing soil quality. We present an organizational framework that attempts first to grade the fitness of the soil for a given use, and then to provide direction toward the detailed processes and attributes that control this fitness. As well, some typical measurements are suggested that, in most cases, are proxies for the processes and attributes.

II. FRAMEWORKS AS TOOLS FOR DIRECTED THOUGHT

A framework is commonly thought of as a structure that gives support and form to a physical construction, such as a building. Scientific frameworks attempt to structure and organize knowledge. Perhaps the most widely used and robust framework in science is the periodic table of elements. However, the periodic table did not systematically and cohesively take shape as knowledge of the chemical world was discovered. Successful organization of elements could begin only after some conceptual tools were formulated to define chemical reactivity. Concepts that linked periodicity to atomic number and valence were pivotal in developing a framework within which elements could be organized and their combinations and reactivity predicted.

In building a framework to organize and rank soil quality, a similar path can be plotted. Extensive physical dissection and observation of soil properties have led to general consensus on the key attributes and processes that are associated with a good quality medium for plant growth. However, the scientific community has yet to decide which conceptual and mathematical tools are needed to formulate a system in which soil attributes can be logically organized and the resulting soil quality assessed or predicted. Perhaps the main reason for this lack of agreement is the complex interdependence of soil attributes and processes at work in a good quality soil. Soil, plants, and climate interact in synergistic and antagonistic ways, with the result that very few soil properties and processes are simply additive or independent. Most natural scientists would agree that nonlinear relationships are normal features in soil–plant systems, arising whenever several factors interact through multiplicative or power functions. Solving these nonlinear feedback equations using chaos theory (Hastings et al., 1993) may be helpful in explaining the complex and interwoven features surrounding the plant–soil ecology.

Soil processes mediate plant growth. Plant growth, in turn, modifies certain soil processes and changes soil attributes, the fundamental components that make up a soil. These components ultimately control the rate of all soil processes. Proxy or surrogate parameters are typically measurements that combine the effects of several processes and attributes. Using this terminology, we built a framework for assessing and predicting the quality of soil for crop production and maintaining adjacent environments. This reductionist approach to categorizing the knowledge of the soil–plant system may appear too restrictive. Nevertheless, it should provide a useful foundation for discussion, recognizing that shortcoming is the necessary forerunner to improvement.

III. DEVELOPING A FUNCTIONAL FRAMEWORK FOR SOIL QUALITY

This framework will provide a uniform structure within which key soil processes and attributes can be organized. This framework is not in itself an evaluation manual. It does not, for example, specify such items as optimum porosity or organic nutrient turnover for a particular kind of use, since such values can never have universal application. Instead, the framework forces the user to rank the adequacy of plant and environmental requirements under the specified conditions and, in so doing, directs the user toward the key soil processes, proxies, and attributes that act to control soil quality or fitness.

A. Soil attributes controlling plant growth

Plant growth depends on the soil's ability to supply water, oxygen, and essential nutrients. Key soil attributes that influence or control these factors are commonly related to each other and are often modified by plant growth. Such highly interconnected systems are not easy to dissect, since removing one factor alters many others. However, from the perspective of plant growth, some useful insights can be drawn from "soil-less" or hydroponic culture. In these systems some type of container is required to hold the nutrients and plant roots, and oxygen and other essential nutrients are dissolved in water and supplied to the roots. Successful plant growth in these systems requires frequent recharge of the nutrient solution with the appropriate balance of essential elements. Soil also provides varying levels of these nutritional and physical requirements. Classifying or quantifying the fitness of a soil to provide these requirements is the basis of a soil quality assessment. Evaluation schemes are given for classifying nutritional (Table 14.1) and physical (Table 14.2) requirements for plant growth. Complex interactions influence the nutritional or physical suitability of a soil. For example, nitrogen (N) supply to the plant depends on the mineralization from the soil organic matter, which depends on the availability of mineralizable substrate and microbial activity, which depends on soil moisture, and so on. All of the soil processes that control the nutritional and physical requirements of plants also vary in response to time and meteorological inputs. As well, spatial variation must be considered if a soil-quality estimate at one location is to be extrapolated to larger areas.

In the framework presented in Table 14.1, plant requirements are grouped according to the soil processes controlling nutrient supply. For example, the

TABLE 14.1

Framework for summarizing a soil's nutritional suitability to support plant growth

Plant requirement	Soil processes	Soil proxies/surrogates (Partial list)	Key soil attributes
Water	• infiltration • retention	• pore size distribution • plant-available water • non-limiting water range	• clay • silt • soil organic carbon
Nitrogen	• microbial growth • respiration • symbiotic fixation	• light fraction • microbial biomass • mineralizable C and/or N	• soil organic carbon • microbial activity • clay • mineralogy
Phosphorus	• mineral weathering/formation • microbial growth • respiration	• microbial biomass • mineralizable C and/or P • soil moisture • pH	• soil organic carbon • microbial activity • clay • mineralogy
Sulfur	• mineral weathering/formation • microbial growth • respiration	• light fraction • microbial biomass • mineralizable C and/or S • electrical conductivity • soil moisture • pH	• soil organic carbon • microbial activity • clay • mineralogy
Potassium Sodium Calcium Magnesium	• mineral weathering/formation • adsorption/desorption	• soil moisture • cation exchange capacity • redox potential • pH	• clay • soil organic carbon • microbial activity • mineralogy
Iron Manganese	• mineral weathering/formation • adsorption/desorption	• redox potential • cation exchange capacity • pH • pore-size distribution	• clay • soil organic carbon • microbial activity • mineralogy
Copper Zinc	• mineral weathering/formation • adsorption/desorption	• cation exchange capacity • pH	• clay • soil organic carbon • microbial activity • mineralogy
Boron Chlorine Molybdenum	• mineral weathering/formation • adsorption/desorption	• electrical conductivity • cation exchange capacity • pH	• clay • soil organic carbon • microbial activity • mineralogy

micronutrient metals iron (Fe) and manganese (Mn) are considered separately from copper (Cu) and zinc (Zn), because pH and Eh are critical to the availability and plant uptake of Fe and Mn. Such groupings are not conclusive, but do provide a basis to simplify and characterize the soil's fitness to supply plant requirements.

TABLE 14.2

Framework for summarizing a soil's physical suitability to support plant growth

Plant requirement	Soil processes	Soil proxies/surrogates (Partial list)	Key soil attributes
Oxygen	• diffusion • root respiration • microbial growth • respiration	• oxygen diffusion rate • pore size distribution • pore continuity • redox potential • non-limiting water range	• clay • soil organic carbon • microbial activity
Rooting space	• shrink/swell • aggregation	• pore size distribution • coefficient of linear expansion • penetrability	• soil organic carbon • clay • mineralogy
Heat	• soil warming	• thermal diffusivity • pore size distribution	• soil organic carbon • clay
Rooting space	• shrink/swell • aggregation	• pore size distribution • coefficient of linear expansion • penetrability	• soil organic carbon • clay • mineralogy
Heat	• soil warming	• thermal diffusivity • pore size distribution	• soil organic carbon • clay

B. Soil attributes controlling environmental requirements or hazards

Soil does more than mediate plant growth. It also absorbs, degrades, and partitions water, gases, and contaminants. Few researchers routinely assess the soil's ability to absorb, degrade, and partition agricultural byproducts along with the soil's fitness for plant production, despite the importance to future plant growth. Table 14.3 summarizes environmental requirements or hazards and systematically addresses the key soil properties and processes. The framework in Table 14.3 groups environmental requirements or hazards as toxins, water, or gases, but these hierarchical groupings may be expanded as other environmental concerns arise.

IV. MONITORING AND PREDICTING SOIL QUALITY

Tables 14.1 through 14.3 form a prototypal framework by which expert knowledge of a specific soil and crop production system can be organized, thereby identifying the key requirements and hazards that limit soil quality. This ranking then directs the user to the key soil processes, proxies, and attributes that control that requirement. The following is a description of how the framework may be applied.

First, bounds are set depending on the scale of the assessment. Regional, local, and site-specific assessments should consider actual plant type(s), as well as

TABLE 14.3

Framework for summarizing a soil's ability to absorb, degrade, and partition toxins, water, and gases

Environmental hazard or requirement transferred through the soil	Soil processes	Soil proxies or surrogates (Partial list)	Soil attributes
Non-nutritional elements (Al^{+3}, Ni, I, Br, F, Cr, Se, Pd, Cd)	• mineral weathering /formation • adsorption /desorption	• toxins • cation exchange capacity • pH	• clay mineralogy • soil organic carbon
Nutritional elements: (excessive nutrients from Table 14.1)	(see Table 14.1)	(see Table 14.1)	(see Table 14.1)
Agricultural by-products: • Pesticides • Residues	• decomposition • adsorption /desorption	• cation exchange capacity • pH	• clay • soil organic carbon
Others......	• erosion • infiltration • drainage	• water-stable aggregates • dispersible clay • K_{sat}[1] • pore size distribution	• soil organic carbon • microbial activity • clay • mineralogy
Flooding and surface runoff	• erosion • infiltration • drainage	• wates • water-stable aggregates • dispersible clay • K_{sat} • pore-size distribution	• soil organic carbon • microbial activity • clay • mineralogy
Greenhouse gases (N_2O, CO_2, CH_4)	• denitrification • respiration • calcite formation • methanogenesis	• gases • acetylene reduction • soil moisture	• clay • soil organic carbon
Other gases (radon, volatile hydrocarbons)			

[1]Saturated hydraulic conductivity.

landscape position and management practices. The fitness of the soil to supply plant nutritional requirements is judged using Table 14.1. The soil's physical suitability to support plant growth is assessed using Table 14.2. A relative ranking of all plant requirements indicates those that are most limiting to plant yield and soil quality.

Selecting the most limiting requirement and reading from right to left gives the soil processes, proxies, and attributes that dominantly control the soil's fitness.

Once the most limiting plant requirements are known, the environmental requirements or hazards can be assessed (Table 14.3). First, the most sensitive adjacent system should be noted. Then the soil's abilities to absorb, degrade, and partition each environmental requirement or hazard are considered, keeping in mind the bounds set for the evaluation. This procedure indicates the quality of the soil in terms of its ability to buffer and regulate environmental hazards and requirements. Reading from left to right in each row of Table 14.3 describes the soil processes, proxies, and attributes that control each environmental requirement.

Considering both the soil's fitness for plant growth and its ability to buffer and preserve sensitive adjacent systems can yield a qualitative assessment of soil fitness, given the boundary conditions. This can be judged by the number of plant-requirement categories receiving acceptable versus low or excess ratings. Changing the boundary conditions results in a different quality assessment for a soil. This approach allows management, crop type, landscape, ecoregion, and adjacent ecosystems to be assessed for their impact on soil quality.

Soil quality can change as a result of human and natural factors. Soil quality is expected to change if the use (crop, management) or sensitive adjacent system changes. If this is the case, then a change in soil quality could be assessed by using Tables 14.1 through 14.3 for a number of scenarios.

V. APPLYING THE FRAMEWORK TO THE PARKLAND REGION OF WESTERN CANADA

A. Need for the framework

Altering management changes the productivity and environmental buffering capacity of the soil. Currently, simulation models are the most flexible tool to assess the impact of management on soil productivity. However, all simulation models are built around a central process or soil function and do not account for other factors that are less relevant to the original model objective. To pick the best simulation model for a given soil–plant system, one must first know the soil processes having the greatest impact on the soil's ability to support that plant.

B. Using the framework

By applying the framework to three soils typical of the dominant soil orders in the parkland region of western Canada (Chernozemic, Luvisolic, and Organic), the soil attributes and processes that are important to productivity and the adjacent environment were identified. Standardized conditions using a reference crop (*Brassica rapa*) with low rates of added fertilizer (25 kg ha^{-1} N; 20 kg ha^{-1} P_2O_5) were specified for each of the example soils. The adjacent ecosystem was defined as having a water table near the surface (10 m), with good connection to a major river system. Using detailed soil data for each genetic horizon of the three dominant soil orders in the parkland region (Table 14.4), soil fitness to supply each plant and environmental requirement listed in column 1 of Tables 14.1–14.3 were rated. Summary of the ratings given by several soil scientists identified the five or six most limiting plant requirements and the single most important environmental concern for each soil. These are described as follows:

1. Chernozem

Given the climate of the parkland region, the Black Chernozemic soil was thought to be most limited by insufficient heat, P, N, S, and water. The capacity to absorb, degrade, and partition environmental hazards was less clear; confusion about the soil's role in protecting the adjacent or sensitive system clouded the summary. However, few negative ratings were given to the soil's ability to buffer or partition environmental hazards or requirements.

2. Luvisolic

The greatest restrictions identified in the Luvisolic soils were N, water, P, S, heat, and rooting space. Low capacities to buffer or partition nutritional elements and water were consistent concerns. Greenhouse gas emissions received scattered rankings.

3. Organic

Restrictions to plant growth on organic soils included N, heat, cation fertility (K), P, and micronutrients. Water storage was not a factor despite the low plant-available water capacity (PAW) of the soil. Environmental buffering and partitioning were generally acceptable, with the exception of greenhouse gas production. Soil

TABLE 14.4

Soil properties[a] of each horizon of a representative soil from the dominant soil orders in the parkland of western Canada

Soil order	Horizon	depth	sand	silt	clay	Org. C	Elec. cond.	CEC	PAW
		cm			%		$mS\ cm^{-1}$	$cmol\ kg^{-1}$	cm
Chernozemic	Ap	0–24	25	48	27	4.0	1	30	36
	Bm	24–49	22	50	28	1.0	1	21	43
	2Ck	49–100	40	30	30	0.3	2	19	71
Luvisolic	Ap	0–13	44	38	18	1.6	0	17	16
	Ae	13–23	44	42	14	0.5	0	12	12
	Bt	23–52	44	28	28	0.5	0	19	38
	Ck	52–100	49	32	19	0.3	0	14	58
Organic	Of	0–20	5	0	0	32.0	0	140	3
	Om	20–40	0	0	0	45.0	0	180	3

[a]Org. C, organic carbon; Elec. cond., electrical conductivity; CEC, cation exchange capacity; PAW, plant available water

properties that control the emission of methane and CO_2 were flagged as important environmental concerns.

Identification of the most limiting plant and environmental requirements provides a clearer view of those soil attributes and processes that limit soil quality in these three soil orders. These soil attributes and processes could be used as key inputs and functions within a simulation model to achieve reasonable predictions of the effect of alternative management on soil quality. Models that account for soil warming, N, P, S, and water will work well in the Chernozemic soils, but may be less accurate in the Luvisolic soils. Crusting and limited rooting space should also be considered in models applied to the Luvisolic soils. Organic soils have some similar limitations, such as N, heat, and P. However, abundance of cations and micronutrients must be strongly considered.

VI. LIMITATIONS AND RECOMMENDATIONS

Currently this framework can serve only to organize and structure an expert's knowledge of soil quality given the specific constraints. Assigning a number to a soil to describe quality at a given time may be feasible if one considers only those attributes that directly affect plant yield. However, if one attempts to evaluate the ability of the soil to absorb, degrade, and partition toxins, water, and gases moving to some adjacent environment, some problems arise. For example, how much value should be assigned to the soil's ability to filter surface water and create a clean pond for waterfowl? How should the soil's value to provide good wildlife habitat be compared with its value for plant production? These questions have a common concern, that is, how all uses and outcomes of a soil system can be normalized and summed. Such fundamental problems will be overcome only by scrutinizing and reorganizing our knowledge of the soil and how it functions. This framework takes a first step toward a standard logical structure for assessing soil quality by clearly stating what limits the soil's quality and how human and natural agents can change these limits.

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Chapter 15

ESTABLISHING A BENCHMARK SYSTEM FOR MONITORING SOIL QUALITY IN CANADA

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I. INTRODUCTION

Water and wind erosion, compaction (destruction of soil structure), salinization, and acidification are natural processes that deteriorate soil quality. Water erosion and salinization are the most significant degradation processes in the prairie region; water erosion, compaction, and acidification are the most important processes in the other agricultural regions of Canada (Coote et al., 1981). These processes are often accelerated by routine agricultural practices such as intensive tillage, application of chemicals, summer-fallowing, and harvesting operations. However, other agricultural activities may reduce or even reverse soil quality decline, including crop rotations (Campbell et al., 1992; Vyn et al., 1992) and conservation tillage (Campbell et al., 1989). Questions about the effects of current farming systems on soil quality arose in the late 1980s during discussions of sustainable agriculture (Bentley and Leskiw, 1984; Poincelot, 1986; Brundtland, 1987). However, the data needed to evaluate these effects were generally not available, or were incomplete or of questionable quality.

In 1988, a team of soil researchers representing all regions of Canada met and agreed that a soil quality monitoring system was needed if the issues of sustainable agriculture were to be properly addressed. They concluded that establishing benchmark sites was the most effective and practical way to collect baseline data sets and to monitor soil quality. Subsequent discussions resulted in developing a hypothesis and objectives, establishing criteria for site selection, determining the number of sites, and identifying appropriate soil properties to monitor. At the time, there existed no national soil quality monitoring system using benchmarks to study. Although the United States monitors about 250,000 sites, no chemical or physical properties were initially measured; only soil type, slope, and land use were noted, and the Universal Soil Loss Equation was applied to estimate current rate of soil erosion. However, in 1989, a proposal for a soil quality monitoring network in France (Lavelle, 1988) suggested establishing 100 sites of 1 ha each. Given the importance of landscape position to soil quality evaluation, (Cao et al., 1994), and the undulating to gently rolling or hummocky terrain of most Canadian agricultural land, it was felt that a benchmark site had to be larger than 1 ha.

In 1989 seven pilot benchmark sites were selected and sampled in eastern Canada. Starting in 1990, three-year funding from the National Soil Conservation Program became available for the benchmark study, and by 1993, an additional 16 benchmark sites were selected and sampled. The collection of baseline data sets for all benchmark sites was completed in 1995. The soil quality monitoring activities will continue for at least 10 years after that time or until trends in soil quality change are demonstrated.

The approach used to monitor soil quality in Canada is best described as "dynamic assessment", a term proposed by Larson and Pierce (1994). Soil quality and the sustainability of a farming system is assessed in terms of the soils performance by measuring attributes of soil quality over time. The purpose of this chapter is to document and discuss the concepts and processes used in establishing a Canadian benchmark network for soil quality monitoring.

II. HYPOTHESIS, OBJECTIVES, AND SITE-SELECTION CRITERIA

A. Hypothesis

Monitoring selected soil variables of a soil landscape under a typical production system for 10 years can demonstrate changes in soil conditions

To test this hypothesis it was necessary to select a typical production system (i.e., a representative soil, landscape, climate, and farm-management system), to establish a baseline data set for all positions of a soil landscape, and to monitor the site for at least 10 years after establishing the baseline data set.

B. Objectives

1. To provide a baseline data set for assessing change in soil quality and productivity (e.g., yields) under representative farm-production systems

A baseline data set is needed to test the hypothesis. For agricultural land, soil quality is closely tied to crop yields and crop quality. Using a representative production system allows broader application of the research findings.

2. To provide a means of testing and validating predictive models of soil degradation and productivity

Predictive models must be tested and validated for Canadian soils and settings. A few years of data collection from benchmark sites may provide a means of evaluating many models for their suitability as predictive tools.

3. To provide a means of evaluating agricultural sustainability of current production systems in major agricultural regions of Canada

Sustainable agriculture is part of the larger objective of sustainable development proposed by the World Commission on Environment and Development (Brundtland, 1987). Sustainable development will remain an important agricultural and environmental issue for many years to come.

4. To provide a network of benchmark sites at which integrated multi-disciplinary research programs can be developed

Detailed "site documentation", in which the site is described and available data sets are listed, will be compiled for every benchmark site. Sites described at this level of detail provide a unique opportunity for integrated, multi-disciplinary research into, for example, developing a better understanding of the processes of soil degradation.

C. Site-selection criteria

Seven criteria for selecting benchmark sites were developed with main goal of selecting typical farm-production systems on dominant landscapes within major agro-ecological regions. Each benchmark site is intended to:

1. Represent a major soil zone and/or agro-ecological region.
2. Represent a typical physiographic region (landscape) and/or broad textural grouping of soils.
3. Represent a major (or potentially major) farm-production system within a region.
4. Complement provincial priorities and opportunities.
5. Provide some potential to be affected by a degradation process(es).
6. Cover about 5 to 10 ha, or a small watershed in some cases.
7. Be limited to cultivated agricultural land.

Top priority was given to the first three criteria. Of the many different agro-ecological areas and farm-production systems that exist in Canada, it was feasible to monitor only a few. However, careful selection of sites according to these criteria allowed the sites to represent the largest possible area and permitted extrapolation of monitoring results and assessments to similar landscapes and production systems over a broader area.

Monitoring activities at benchmark sites can complement provincial research. Sites that could provide multiple benefit or lead to collaboration in research and monitoring activities were considered highly desirable. Other benchmark sites were located at established Research Centres, such as are operated by Agriculture and

Agri-Food Canada, to form a monitoring network to extend existing programs and to make use of long-term soil, climate, and yield data already collected in their agro-ecological areas.

Selected sites have the potential to be affected by a degradation process or processes, that is, have the possibility of change in one or more soil characteristics because of management. Although most change is negative (e.g., loss of organic matter and nutrients resulting from erosion), there may be situations in which change is positive (e.g., an increase in soil organic matter content). Representative agricultural land of reasonably good quality was usually selected. Although areas with very degraded soils could have been selected, they were generally avoided, usually because they were not representative of a region. Information about very degraded soils tends to be plentiful; much less is known about soil quality change on medium- to good-quality farmlands that dominate many agricultural regions.

An area of 5 to 10 ha, or a small watershed in some cases was considered an appropriate size for benchmark monitoring for three reasons:

- All segments of the targeted landscape could be sufficiently represented, especially for replication of sampling positions.
- This area is sufficiently large to not interfere with most farm management systems.
- This area is sufficiently small to be suitably characterized in detail and monitored by the benchmark-site manager.

Each benchmark site is restricted to cultivated agricultural land. Since there were only enough resources to establish a few sites, cultivated landscapes were emphasized. Although rangelands were recognized as important in the provinces of Alberta, Saskatchewan, and British Columbia, insufficient resources and expertise excluded such landscapes from the benchmark-site study. The benchmark-site study methodology can be extended to rangelands at some later date.

Final site selection occurred in the field. Factors that affected the final decision included representativeness of the soils and topography, type of farming system in use, cooperativeness of the farm operator, and, in some cases, proximity to a climate station. According to these site selection criteria, 25 sites were identified and a total of 23 sites were selected and sampled by 1992. The geographic distribution of the 23 benchmark sites is shown in Fig. 15.1, and a brief description of the sites is provided in Tables 15.1 and 15.2.

III. BASELINE DATA SETS

A. Farm history

A farm history database was recorded based on an interview with the farmer. The farm history database has three parts:

1. Site identification, including the site identification number, site name, legal location, agro-ecoregion, site manager's name, and farmer's name, address, and telephone number.

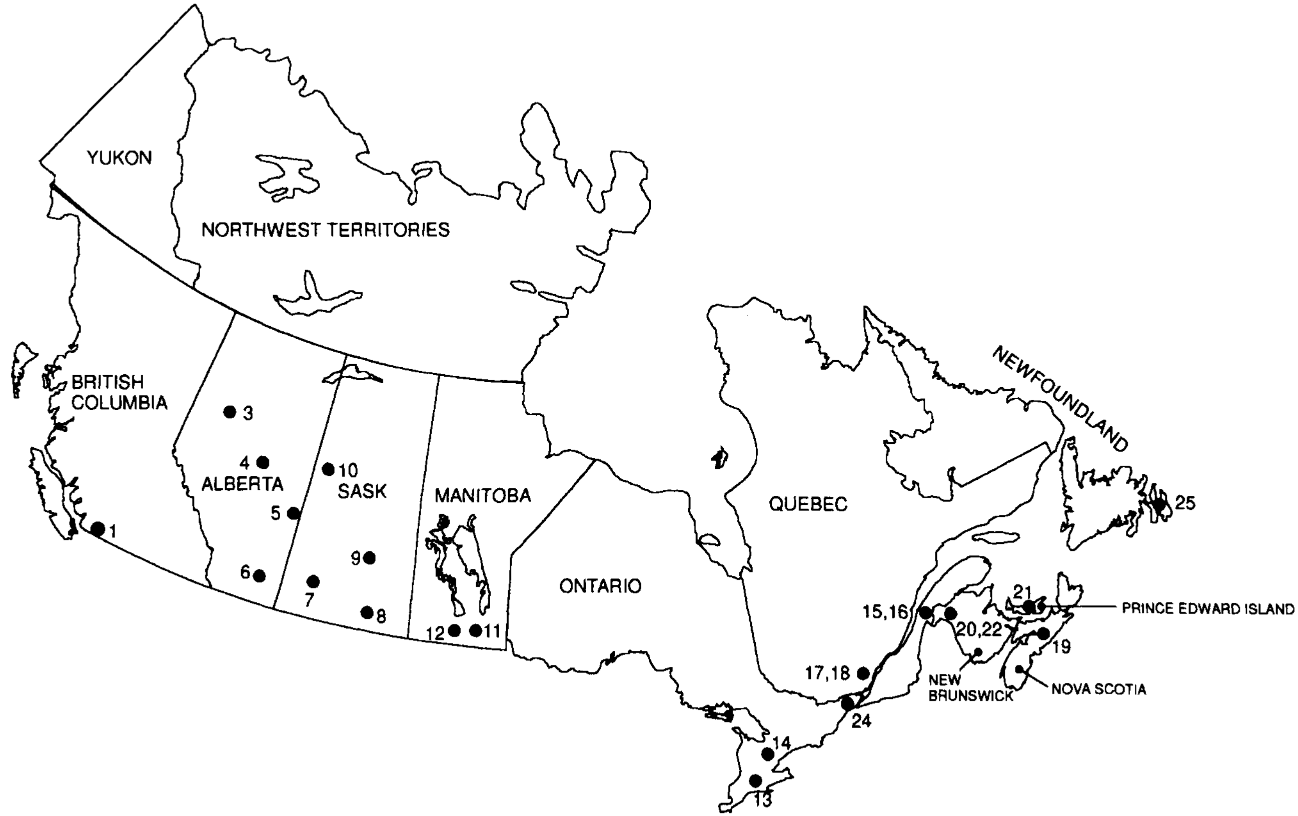

Fig. 15.1. Location of the benchmark sites in Canada.

TABLE 15.1

Site name, location, and ecoregion for 23 benchmark sites in Canada

Site no.	Site name	Location	Ecoregion[1]	Size (ha)	Year established
01-BC	Chilliwack	49° 09′N 121° 53′W	Lower mainland	5.0	1991
03-AB	Falher	55° 43′N 117° 17′W	Peace lowland	7.5	1991
04-AB	Mundare	53° 33′N 112° 17′W	Boreal transition	16.3	1992
05-AB	Provost	52° 25′N 110° 07′W	Aspen parkland	8.8	1990
06-AB	Bow Island	49° 42′N 111° 26′W	Mixed grassland	7.5	1991
07-SK	Swift Current	50° 15′N 107° 43′W	Mixed grassland	4.8	1992
08-SK	Regina	50° 18′N 104° 32′W	Mixed grassland	7.5	1991
09-SK	Termuende Farm	51° 52′N 104° 51′W	Aspen parkland	7.3	1990
10-SK	Loon Lake	54° 03′N 109° 05′W	Boreal transition	7.5	1991
11-MB	Brunkild	49° 30′N 97° 34′W	Aspen parkland	5.1	1991
12-MB	Carman	49° 28′N 98° 03′W	Aspen parkland	5.1	1990
13-ON	Ingersoll	43° 00′N 80° 48′W	Lake Erie lowland	4.0	1990
14-ON	Rockwood	43° 38′N 80° 11′W	Manitoulin–Lake Simcoe	1.5	1991
15-QU	St-Elzear	46° 25′N 71° 07′W	Appalachians	5.2	1989
16-QU	St-Elzear	46° 25′N 71° 07′W	Appalachians	5.2	1989
17-QU	St-Marc	45° 38′N 73° 13′W	St-Laurent lowland	5.1	1989
18-QU	St-Antoine	45° 39′N 73° 10′W	St-Laurent lowland	4.3	1989
19-NS	Stewiacke East	45° 10′N 63° 18′W	Annapolis–Mines lowland	4.8	1989
20-NB	Drummond	47° 00′N 67° 41′W	Saint John river valley	4.3	1989
21-PE	Albany	46° 17′N 63° 37′W	Prince Edward island	7.5	1989
22-NB	Salmonhurst Corner	46° 59′N 67° 40′W	Saint John river valley	8.3	1990
24-ON	C.E.F.	45° 22′N 75° 44′W	St-Laurent lowland	2.5	1991
25-NF	St. John's Res. St.	47° 31′N 52° 47′W	Maritime barrens	5.0	1992

[1] Ecological Stratification Working Group (1995).

TABLE 15.2

Soil, landform, and management system of benchmark sites in Canada

Site no.	Parent material and surface form	Soil great group[1]	Soil series	Problem	Cropping system	Tillage system
01	Medium textured fluvial Level	Humic gleysol	Pelly	Water quality, compaction	Silage corn	Tiled drainage, conventional tillage
03	Fine textured glaciolacustrine Level	Dark gray	Falher	Structure, O.M.[3] degradation, acidity	Cereals – canola – forage	Conventional tillage
04	Medium textured fluvial over till Undulating to ridge	Black	Hobbema	Water & wind erosion, O.M.[3] loss, salinity	Cereals – forage	Conventional vs no-till
05	Medium textured fluvial over till. Hummocky to undulating	Dark brown	Hudhenden & Provost	Wind & mechanical erosion, O.M.[3] degradation	Canola – wheat – fallow	Conventional tillage
06	Medium textured lacustrine over till. Undulating	Brown	Cranford & Helmsdale	Salinity, wind erosion O.M.[3] loss, structure	Wheat (seed) – beans – sugar beets	Irrigated conventional tillage
07	Medium textured loss over till Undulating, dissected	Brown	Swinton[2]	Wind erosion, water erosion	Wheat – fallow	Conventional tillage
08	Fine textured lacustrine Undulating	Dark brown	Regina[2]	Wind erosion	Extended rotation, mainly wheat	Conventional tillage
09	Medium textured till Undulating	Black	Oxbow[2]	Water & wind erosion	Extended rotation (continuous cereal)	Conventional tillage
10	Medium textured till Undulating	Gray luvisol	Loon river[2]	Water & wind erosion, structure	Barley – oilseed	Conventional tillage
11	Fine textured lacustrine Level	Humic gleysol	Osborne & Glenmor	Wind erosion, O.M.[3] degradation, salinity	Continuous cereals	No-till
12	Medium over fine textured lacustrine. Level	Black	Rignold & Kronstal	Wind erosion, O.M.[3] degradation	Wheat – canola	Minimum tillage
13	Fine textured glaciolacustrine Level and hummocky	Gray brown luvisol	Huron & Perth – Brookston	Structure, water erosion	Corn – soybean – wheat	Minimum tillage

[1]Agriculture Canada Expert Committee on Soil survey (1987).
[2]Soil association.
[3]Organic matter.

TABLE 15.2 (continued)

Site no.	Parent material and murface form	Soil great group[1]	Soil series	Problem	Cropping system	Tillage system
14	Medium textured till. Rolling	Gray brown luvisol	Guelph & London – Parkhill	Water erosion, compaction	Corn – soybean – wheat	No-till
15	Medium textured till. Rolling	Dystric brunisol	Woodbridge	Water erosion, O.M.[3] degradation	Silage corn – forage	Conventional tillage
16	Medium textured till. Rolling	Dystric brunisol	Woodbridge	Water erosion, O.M.[3] degradation	Silage corn – forage	Conventional tillage
17	Marine clay. Level	Humic gleysol	Providence	Compaction, O.M.[3] degradation	Corn – forage	Conventional tillage
18	Marine clay. Level	Humic gleysol	Providence	Compaction, O.M.[3] degradation	Corn – wheat – soybean – barley	Minimum tillage
19	Medium textured till. Undulating	Gray luvisol	Queens	Compaction, water erosion	Corn – forage	Moldboard plow, Spring disked
20	Corn textured till. Rolling	Humo-ferric podzol	Holmesville	Water erosion, compaction	Potato – potato – grain	Chisel plow
21	Coarse textured till. Undulating to rolling	Humo-ferric podzol	Charlottetown	Unstable structure, compaction, water erosion	Potato – grain – forage – forage	Conventional (plowdown forage)
22	Coarse textured till. Rolling. Diversions and grassed waterway	Humo-ferric podzol	Holmesville, Undine	Water erosion, compaction	Potato – potato – grain	Chisel plow
24	Medium textured fluvial. Level	Humic gleysol	North Gower	Organic matter degradation, compaction	Corn – soybean – alfalfa	Conventional & no-till
25	Medium textured fluvial. Level	Humic gleysol	North Gower	Organic matter degradation, compaction	Corn – soybean – alfalfa	Conventional & no-till

[1] Agriculture Canada Expert Committee on Soil survey (1987).
[3] Organic matter.

2. Site history, including land-acquisition date; first cultivation date; early years' land management; major changes in agronomic practices; crop rotation; tillage system; crop yields and quality; commercial fertilizers, organic fertilizers and soil conditioners; chemical pesticides/herbicides; and soil degradation problems.
3. Current cropping and tillage practices, including crop rotation system; tillage, crop management, and harvesting procedures; farm-machinery inventory; and special notes. An example of a farm history database is given by Wang et al. (1994).

B. Soil map

A soil map at a scale of 1:2000 or larger was prepared for each site. This usually consisted of single soil-series map units with surface-texture phases. It was compiled from at least 40 ground inspection points.

C. Contour map

Topographic data are a key element in the characterization of benchmark sites. This information can be used to display landscape features and to provide the basis for overlaying other soil and terrain characteristics. Locations of sampling points and other features can be pinpointed for future repositioning. A detailed contour map, plotted at 0.2 to 0.5 m intervals depending on local relief, was created for each site. Two permanent topographic marks were installed at each site for future reference.

D. Soil sampling and preparation

1. Representative pedons

The procedures for taking undisturbed cores and loose soil samples from representative pedons were as follows:

- Two pedons that represented typical soils of the benchmark site were selected.
- A pit about 1 m wide by 2 m long by 1.5 m deep was opened at the selected locations.
- All main soil horizons on one exposed face of the freshly dug pedon were identified and described according to Day (1982).
- Cores of 7.5-cm diameter by 7.5-cm length were collected from each horizon using a hand-operated Uhland sampler (McKeague, 1978). In general, five cores were taken from Ap horizons, three cores from other horizons.
- About 1 kg of representative loose soil was collected from each horizon.

2. Loose samples for baseline data set

Since most changes in soil properties occur in the surface layer, sampling was concentrated in the Ap horizon, with occasional samples taken from B and C horizons. From 60 to 100 sampling points were selected to characterize a benchmark site, using one of the following two sampling methods:

- Grid sampling: a 25×25 m grid was generally used to cover an entire benchmark site, typically resulting in about 100 grid sampling points. The grid-sampling method was used on benchmark sites that either have no significant surface relief or are gently rolling or undulating. Permanent grid baselines were established to allow for exact location of grid coordinates for resampling and *in situ* measurements.
- Transect sampling: This is a stratified random sampling method, as described by Wang (1982), in which landforms were typically used for stratification. This method was used on benchmark sites with a hummocky surface relief and typically used five to eight transects (from crest to toe of the slope), for a total of about 60 sampling points on each benchmark site. Because of stratification, fewer sampling points were needed for the transect method than the grid method. Permanent transect baselines were established to allow for exact location of transect coordinates for resampling and *in situ* measurements.

A representative loose sample of the upper 15 cm of the Ap horizon, or the whole Ap in some cases, was taken at every sampling point. Additional loose samples were collected at a depth of 50–60 cm (usually the B horizon) at every fourth sampling point; at a few sites, a C horizon sample at a depth of about 1 m was also collected. For each sample, depth of sampling, soil colour, structure, field texture, consistence, and landscape position were recorded.

3. Loose samples for ^{137}Cs analysis

Loose samples were collected for ^{137}Cs analysis at selected benchmark sites where surface-soil redistribution or water erosion are being monitored. The sampling method was the same as that for collecting loose samples for baseline data except that only the Ap (or Ah) horizons was sampled at every sampling point. Bulk density samples, collected in 7.5×5.0 cm Kubiena boxes or 5 cm $\times$ 5 cm cores, were taken from the middle of the A horizon at every sampling point. The depth of the A horizon was recorded.

4. Sample handling, preparation, and archiving

Core samples were stored at 4°C in a cold room until processed. Loose samples were air-dried and ground to separate the fine-earth fraction (<2 mm) from coarse fragments. The fine-earth fraction of the loose sample was split into two equal parts. One part was used for detailed laboratory characterization. The other part was stored in a paper container lined with plastic and was archived for future use.

E. Laboratory Characterization

Various chemical, physical, and mineralogical properties of soil deemed to be important in soil quality were analyzed to establish baseline data sets for all benchmark sites. Soil properties (mostly of Ap horizons) were classified into one of the following three categories (Table 15.3):

TABLE 15.3

Soil properties (chemical, physical, biological, and mineralogical) measured at benchmark sites and monitoring frequency

Sensitive properties[1]	*Nonsensitive properties*[4]
Soil reaction (pH)	Particle-size distribution
Available phosphorus and potassium	Clay mineralogy[5]
Organic carbon	Total surface area
Total nitrogen	Total elements (aluminium, calcium, cobalt,
Bulk density	chromium, copper, iron, potassium, lithium,
Dry-aggregate size distribution	magnesium, sodium, nickel, lead, zinc)
^{137}Cesium distribution	
Extractable iron and aluminium[2]	
Moderately sensitive properties[3]	*Properties measured in situ*[6]
Cation exchange capacity	Saturated hydraulic conductivity
and exchangeable cations	Near-saturated hydraulic conductivity
Carbonates	Penetrometer reading and soil moisture
Soil moisture retention	Electromagnetic ground conductivity[7]
	Biopore and root counts
	Earthworm counts[8]
	Crop yields

[1] Measured every few years.
[2] For Podzolic soils only.
[3] Measured every 10 years.
[4] Measured only at the beginning of the observation period to establish baseline data.
[5] Heavy application of nitrogen and potassium fertilizer may alter some silicate clays and special studies may be needed.
[6] Measured in the field annually.
[7] Only in areas with potential salinity problems.
[8] Except in the Prairie Provinces.

1. Sensitive properties, which are likely to change significantly in less than 10 years, including pH, available P and K, organic C, total N, ^{137}Cs distribution, bulk density, and dry aggregate size distribution.
2. Moderately sensitive properties, which are likely to change in decades, including exchangeable cations, cation exchange capacity (CEC), carbonates, and soil moisture retention.
3. Non-sensitive properties, which are not expected to change significantly in 100 years, including particle-size distribution, clay mineralogy, total surface area, and total elements (Al, Ca, Co, Cr, Cu, Fe, K, Li, Mg, Mn, Na, Ni, Pb, and Zn).

The non-sensitive properties, although not expected to change significantly in the duration of this monitoring study, are important properties in assessing the overall soil quality of each benchmark site. However, in the presence of severe erosion (e.g., such as at Site 20) non-sensitive properties may change significantly in a few decades and should also be monitored. Larson and Pierce (1994) divided dimensions of soil quality into two categories, inherent and dynamic. The inherent dimensions of soil quality are equivalent to the non-sensitive category, and the dynamic dimensions of soil quality are equivalent to the sensitive and moderately sensitive categories.

Measurements of soil-moisture retention and bulk density were completed for all core samples taken from the representative pedons of each benchmark site. Additional bulk density determinations were also made for samples taken in Kubiena boxes. ^{137}Cs was measured for all samples taken for ^{137}Cs studies (at Sites 5, 9, 14, 15, 16, 19, 20, 21, and 22). Extractable Fe and Al were determined for all loose soil samples from Quebec and the Atlantic Provinces. The rest of the sensitive and moderately sensitive properties were determined for all loose samples from all benchmark sites. The non-sensitive properties were determined for about 15% of the loose samples selected from each benchmark site.

The laboratory methods used are described by Wang et al. (1994). Specific procedures and methodologies were recorded to ensure consistency in future analyses after resampling. Stringent quality-control methods (Sheldrick, 1986) were implemented.

F. Field data collection

Measurements of selected physical and biological properties of soil were also made *in situ* at most of the benchmark sites. Physical properties measured included saturated hydraulic conductivity, near-saturated hydraulic conductivity, penetrometer readings with soil moisture data, and electromagnetic ground conductivity (EM 38). The two hydraulic conductivity measurements will provide information on soil water movement within the rooting zone under both saturated and unsaturated conditions. For long-term monitoring, hydraulic conductivities will also provide information on changes in soil structure and the rate of structural degradation or aggradation. Penetrometer readings provide information on soil strength (a major factor influencing plant-root growth), which in turn determines, to a large degree, the amount of soil moisture actually available to plants.

Biological properties observed include biopore counts, counts of earthworms (where present), and crop yields. Biopores, which include root channels and earthworm holes, are important to saturated hydraulic conductivity and aeration of soil. In humid regions, a high earthworm population provides more desirable soil structure, as well as readily available organic nutrients. Crop yield, including both quantity and quality of the crop, is a major indicator of soil quality.

Climate is the most important factor limiting agriculture in Canada. For most benchmark sites, climatic information was collected from a nearby weather station(s) operated by Environment Canada. Information typically includes daily mean soil temperature at a depth of 50 cm, mean daily aerial temperature, degree days, frost free days, daily precipitation, hours of sunlight, etc. Automated weather stations (Campbell Scientific CR 10) were installed in eight of the 23 benchmark sites.

Methods for field data collection are given in detail by Wang et al. (1994).

IV. MONITORING FREQUENCY AND RESAMPLING

Monitoring frequency is determined by the rates at which soil properties of concern change. For the properties to be characterized in the laboratory, resampling will be required. In principle, the same number of sampling points and sampling coordinates as for the baseline data are recommended for resampling. This resampling design will fix all external variables except time. The same principle also applies to monitoring properties, measured *in situ*; these measurements should be conducted at approximately the same time each year.

It is recommended that sensitive soil properties be monitored about every five years (Table 15.3). The exact frequency for resampling is dictated by the rotation system for each site. Resampling should be carried out under the same crop and time of year as when baseline sampling took place. Moderately sensitive soil properties should be monitored about every 10 years. Non-sensitive soil properties do not require monitoring. In some special cases, heavy application of ammonium and potassium fertilizer may alter some silicates such as expandable clay minerals in just a few years (Ross et al., 1985, 1989), and special monitoring studies may be needed.

Data collected *in situ*, such as hydraulic conductivities, penetrometer readings and soil moisture, biopores, earthworm counts, and EM 38 conductivity measurements, should be measured yearly for at least five years or for a complete rotation cycle, whichever is longer. After completing the initial five years of yearly data collection, these *in situ* measurements will be monitored every five years. Information such as crop yields and the farmer's management diary is collected annually. Climatic data is automatically recorded hourly and daily.

V. BENCHMARK SITE DATABASE SYSTEM

A relational database was designed for the benchmark site study. Given the host of data types for a variety of measured properties, the main goal was to attain efficient data storage that would allow reasonably simple manipulation and retrieval. This was achieved by using many small files, developed in dBASE IV (Ver. 1.5). Each file contains similar types of data on similar kinds of soil and landscapes. Most files can be linked to perform analyses across data types and soil landscapes. Currently the files contain baseline, reference, or current data. Results of resampling measurements will be entered in files similar to those containing baseline data so that temporal comparisons can be made.

VI. GENERAL COMMENTS

It has been more than five years since the first benchmark site was selected and sampled in 1989. Early results of the program allow some conclusions about effects of farming practices on soil quality (Wang et al., 1995). The hypothesis, objectives, site-selection criteria, and the overall plan of the monitoring system remain satisfactory. The following modifications of the monitoring program are being implemented:

- *In minimum till or no-till systems, sampling the Ap horizon in two or three 5-cm increments.* Conservation tillage, in general, has its greatest effects on the upper Ap horizon. The effect of tillage practices on soil quality may be easier to detect if Ap horizons are sampled in 5-cm increments instead of the whole depth of the Ap.
- *Monitoring organic matter qualitatively as well as quantitatively.* The link between soil organic matter quality and soil quality has been increasingly recognized in recent years. There is a need to monitor soil organic matter quality in addition to quantity (Gregorich et al., 1994). For example, recommended soil organic matter characterization should also include the light fraction (Gregorich and Ellert, 1993) and soil microbial biomass (Voroney et al., 1993).
- *Monitoring groundwater table if it is present within 1.5 m of the soil surface.* The use of agricultural chemicals and their effect on groundwater quality are a serious environmental concern (Bouwer, 1989; Logan, 1990).

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Chapter 16

CASE STUDY OF SOIL QUALITY IN SOUTH-EASTERN AUSTRALIA: MANAGEMENT OF STRUCTURE FOR ROOTS IN DUPLEX SOILS

B. COCKROFT and K.A. OLSSON

I. INTRODUCTION

This chapter concerns those aspects of soil structural quality that determine productivity of irrigated crops in duplex soils (Alfisols) in the semi-arid environment of Victoria in south-eastern Australia. A duplex soil typically shows a texture contrast between the A and B horizons comprising the solum. Crop productivity in duplex soils is well below the potential set by solar radiation and water supply (Cockroft and Martin, 1981; French and Schultz, 1984). In Victorian duplex soils, ideal structure is often rare in the field because of compaction by traffic, crusting, hard-setting, slumping, natural pans, and the coalescence of even the most stable aggregates. Even well prepared seedbeds usually lose porosity and increase in strength within a few weeks of sowing the crop; below the seedbed and in non-tilled soils such as in orchards, the soil structure remains poor.

Improving the structure of a cropped soil to a defined ideal requires an understanding of processes in structure formation and stabilization (Greenland, 1981; Dexter, 1988), from clay aggregates to macroaggregates, that occur in alfisols (Oades and Waters, 1991). Systems of soil management have been developed for duplex soils in Victoria that improve and manage soil structure so as to increase crop yields towards the potential. The objectives of this chapter are to describe the soil quality attributes needed for best root growth and functioning, and how to maintain the required structure, using information derived mainly from local studies. These approaches are illustrated using examples drawn from experience with irrigated tree and row crops.

II. SOIL QUALITY FOR ROOT GROWTH AND FUNCTION

The functions of roots in crop productivity include the uptake of water and nutrients, support for the plant, the storage of assimilates, and control over the growth of the aerial parts of the plant (Thornley, 1977; Blaikie and Mason, 1993). Roots are able to sense conditions in soil and communicate signals to the shoots, where they influence stomatal conductance and growth (Passioura, 1991). When roots are confined by dense soil, the shoots slow their growth even where supplies of water, oxygen, and nutrients are available to the roots (Masle and Passioura, 1987). Table 16.1 summarizes the main physical attributes for optimum root growth and function in a duplex soil. Although these are based on topsoil data and are clearly a simplification, they should also guide the restructuring of B horizons.

In irrigated duplex soils, it is important to improve the soil physical properties (listed in Table 16.1) throughout the solum to about 500-mm depth. Penetrometer resistance should remain below 0.5 MPa as the soil dries (Cockroft et al., 1969). In Figure 16.1 the penetrometer resistance–water content curves of the A horizons from two non-tilled orchard soils are compared. Both soils are water stable and high in macroporosity. Soil A is a typical A horizon in which the penetrometer resistance increases to > 3 MPa within two to three days following irrigation or rain. Roots of fruit trees grow little during the irrigation season on these soils (Cockroft and Olsson, 1972). Curve B shows the strength curve from a unique duplex soil in which the penetrometer resistance is almost independent of soil water content on drying over the range of matric suction to 1.5 MPa. Although we do not yet know why this duplex soil is soft and loose, it does show that it is possible to manage these soils to maintain the ideal mechanical resistance.

TABLE 16.1

Soil physical attributes for optimum root growth and function in duplex soils

Soil quality attribute	Physical measurement	Critical level or range	Reference
Soil volume	Depth	500 mm	Greenland (1981)
Mechanical resistance	Penetrometer resistance	< 0.5 MPa	Cockroft et al. (1969)
Soil water	Macropores (> 30 μm)	> 15%	Hall et al. (1977)
	Storage pores (30–0.2 μm)	> 20%	
	Flow to root	Hydraulic cond. at 100 kPa suction > 10^{-4} m day^{-1}	
Aeration	Air-filled porosity after 24 h drainage	> 15%	Cockroft and Tisdall (1978)
	O_2 content of soil air	> 10%	Dexter (1988)
Temperature		18–28 °C	Shaw (1952)

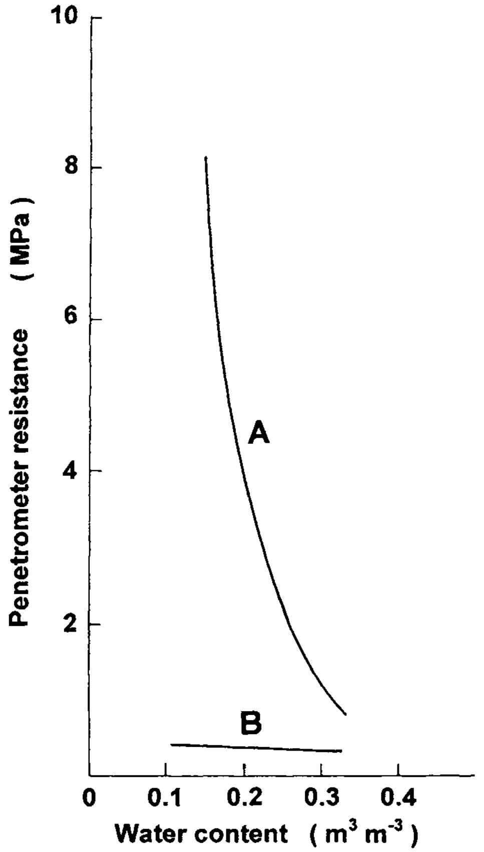

Fig. 16.1. Penetrometer resistance as a function of water content in non-tilled A horizons from two orchards on duplex soils. Curve A represents a typical soil. Curve B is a unique soil beneath a 40 year old pear orchard. (From Cockroft, B., Cass, A., Lanyon, D. and Olsson, K.A., unpublished data).

Roots need both a rapid infiltration and redistribution of water at irrigation and an adequate store of soil water. Thus, macropores should occupy > 15% and storage pores > 20% of the soil volume (Hall et al., 1977). For duplex soils, it is suggested that the size distribution of these pores should provide an unsaturated hydraulic conductivity of $> 10^{-4}$ m day^{-1} at 100 kPa matric suction to enable optimum water flow to the root. Hydraulic conductivity characteristics are shown schematically in Figure 16.2, representing a highly conductive soil, C, in contrast to the A and B horizons of a typical duplex soil.

For adequate aeration it is necessary for the soil to have a macroporosity of 15% continuous throughout the solum to ensure rapid drainage. An air-filled porosity of 10% 24 h after irrigation is usually considered sufficient (Wesseling, 1974; Dexter, 1988), but > 15% is more appropriate in soils high in biological activity (Cockroft

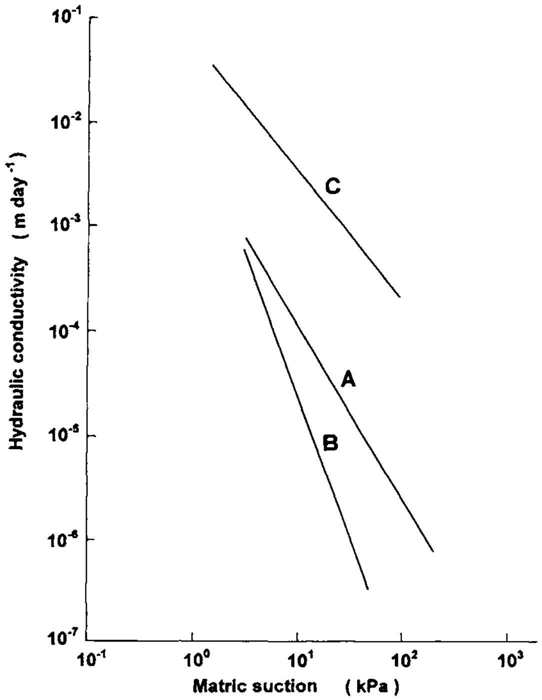

Fig. 16.2. Schematic comparison of the unsaturated hydraulic conductivity characteristics for the A and B horizons (A and B respectively) of a duplex soil and for a highly conductive field soil (C).

and Tisdall, 1978). The soil air should contain at least 10% oxygen for optimum microbial activity (Dexter, 1988).

III. STRUCTURE FORMATION AND STABILIZATION

Olsson et al. (1995) describe the production and stabilization of structure for roots in duplex soils—specifically, how aggregates are built up from smaller particles, and how fragments of tilled dense soil could be converted into soft, porous aggregates. Table 16.2 brings together the main mechanisms involved in forming and stabilizing aggregates at each level in the hierarchy and the implications of these for managing duplex soils. Tisdall and Oades (1982) and Oades (1984) describe the processes involved in the aggregation of Alfisols, including the importance of organic matter to both macroaggregate and microaggregate formation and the role of gypsum and lime. The latter amendments provide soil solution Ca to help flocculate colloids and provide cation bridges between humic materials and clays (Emerson and Greenland, 1990).

Soft, porous aggregates can also be developed from the mellowing of dense soil. This requires fragmentation of the dense layer by tillage, the penetration of the fragments by roots and fungal hyphae, loosening of the fragment surface by roots, rupture during wet/dry cycles, increases in organic matter from roots and infusion of Ca from gypsum (Olsson et al., 1995).

TABLE 16.2

Important processes in the formation and stabilization of aggregates in Australian duplex soils

Aggregate	Size range (µm)	Formation and bonding agents	Implications for management
Clay aggregates	< 2	Clay particles condense to form domains and quasi-crystals bonded by electro-static forces (Emerson, 1983)	Calcium ions and electrolyte needed, both from gypsum (Rengasamy et al., 1984)
Microaggregates	2–250	Clay aggregates sorbed onto organic fragments stabilized by microbes; bonded by organo-mineral complexes (Oades and Waters, 1991)	Need to build up organic matter and calcium (Emerson and Greenland, 1990), and rhizosphere (Oades and Waters, 1991). Avoid compressive shear (McGarry, 1989)
Macroaggregates	> 250	Roots and fungal hyphae link microaggregates (Tisdall and Oades, 1982). Biological agents mellow and give porosity and resistance to compaction (Dexter, 1988; Emerson and Greenland, 1990). Fungal and bacterial mucilages act as glues (Foster, 1994), and inorganic materials act as cements (Oades, 1984)	Build up organic matter and calcium. Maximize root and hyphae growth and minimize tillage (Oades, 1984). Manage water for slow wetting and good drainage (Olsson et al., 1995). Dedicate traffic to lanes to reduce compaction (Soane et al., 1981)

IV. MAINTENANCE OF STRUCTURAL QUALITY

This section outlines the steps required to maintain structural quality in some Australian duplex soils used for irrigated fruit crops and row crops. The system, which is a form of ridge tillage (Cockroft and Tisdall, 1978), aims at improving attributes of soil structure for better growth and functioning of the crop roots in the surface soil and subsoil (Table 16.3). In several commercial orchards the new system has produced yield increases from an average of 30 t ha^{-1} in conventional systems to 60 t ha^{-1}, with the highest yields approaching 100 t ha^{-1}.

TABLE 16.3

Soil management needed to achieve optimum soil physical quality in duplex soils

Soil quality attribute	Soil conditions needed for roots	Operations in the field
Volume	• Solum loosened to 500 mm • Maximum volume of surface soil • Separation from traffic • Minimum cutting of roots by cultivation	• Deep till • Form surface soil into beds • Confine traffic to lanes • Reduce tillage operations
Mechanical resistance	• Loose, soft soil • Low strength aggregates • No clods	• All of the above • Ensure high Ca and OM • Winter cover crops • Wet slowly and avoid saturation
Soil water	• Adequate macropores • Adequate storage pores • Moist soil/reduced evaporation	• All of the above • Summer dry grass mulch • Maintain moistness
Aeration	• Adequate macropores • Remove excess water • Allow deep percolation • No water table • No soil saturation	• Grade land surface • Deep till to permeable layer • Form beds or hills • Irrigate precisely • Use winter crops for soil drying • Use test wells
Temperature	• Optimum 18–28 °C • Minimum 10 °C • Maximum 35 °C (depending on species)	• Encourage shade from canopy • Provide a mulch

A. Deciduous orchards

The system of soil preparation and management here described is a major development of that described by Cockroft and Tisdall (1978) for fruit trees on poorly structured duplex soils under irrigation in the Goulburn Valley, Victoria, situated in temperate south-eastern Australia. The soil management system aims to improve soil structural quality based on the information given in Tables 16.1 and 16.2. Some of these features are shown in Figure 16.3. The soils have a 150-mm depth of loam overlying a red-brown clay (Stace et al., 1968). The surface soil is high in silt and fine sand, and its structure rapidly deteriorates when cultivated and when the organic carbon declines to $< 1\%$. The clay subsoil (B horizon) has a saturated hydraulic conductivity of < 30 mm day^{-1} and a macroporosity $< 5\%$, so that profile drainage is poor; the C horizon at approximately 500-mm depth is more permeable

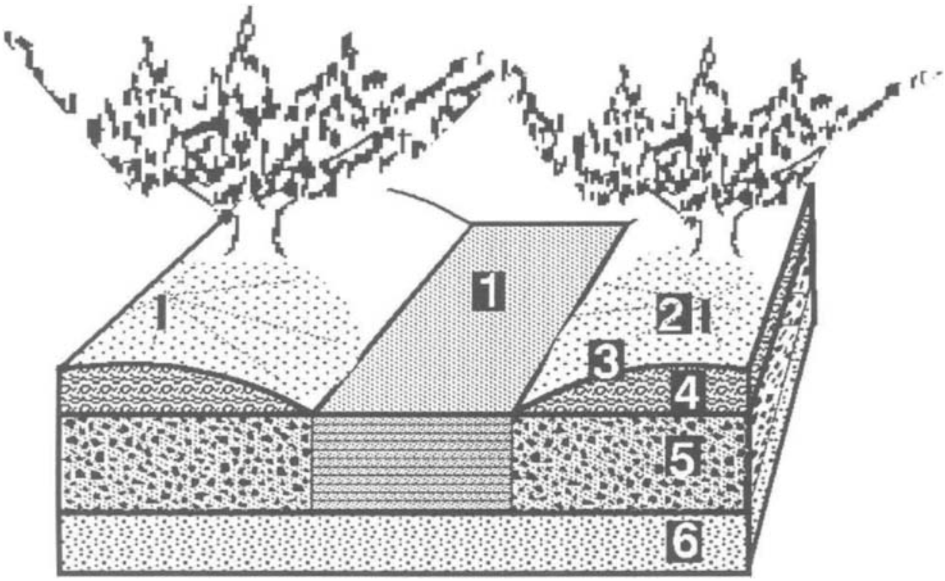

Fig. 16.3. System of soil management for a deciduous orchard on a duplex soil. 1) Dedicated traffic lane, 2) Irrigation by fine sprinklers, 3) Ryegrass (not shown) grown in autumn and winter (the sprayed off grass becomes a mulch in spring and summer), 4) Hilled beds of topsoil, 5) Loosened B horizon beneath beds, 6) Permeable C horizon at 500 mm depth.

(>60 mm day^{-1}). Both the A and B horizons show substantial increases in mechanical resistance as they dry (Fig. 16.1).

The procedure for setting up the system starts with an initial survey of the soil to indicate the depth to restructure the subsoil; textures; structure; consistence; permeabilities; requirement for gypsum, lime and phosphorus; and any need for leaching. If the soil is high in organic matter ($>2\%$), subsequent maintenance of structure will be much easier. The land surface is graded for good surface drainage (slope of $\geq$1:800). The solum is loosened by deep tillage prior to bed formation in the surface soil. The grower does not cultivate the soil again throughout the life of the trees. Once the beds are formed, the grower immediately sows a grass, preferably ryegrass (*Lolium* sp.), and grows it through the winter to reduce hard-setting and coalescence of the loose soil. Each subsequent autumn, ryegrass grows as a volunteer. In the spring the grower kills the grass with weedicide, retaining the dry grass on the bed to provide a summer mulch, and controls any further weed competition until autumn. The grower installs irrigation sprinklers along each row that apply small droplets of water slowly (at a rate of no more than 6 mm h^{-1}) and uniformly along the bed. In heavy rain, the tree canopy, the mulch, and the lateral slope of the bed help protect the soil. The grower irrigates and fertilizes for maximum yields by keeping the soil wetter than 100 kPa suction and leaf nutrients at recommended levels.

Aspects of soil management that improve and maintain soil structural quality for optimum root growth and functioning are described in the subsections below.

1. Increasing soil volume

The volume of soil with the structure that meets the specifications for root function set out in Table 16.1 is very small in conventional orchards. The new system increases soil volume for tree roots by forming up surface soil from the traffic lanes into beds and by confining traffic to the lanes between the beds. Richards and Cockroft (1974) showed that in a non-cultivated peach orchard, root concentrations averaged 20 km m^{-3} of soil compared to zero in cultivated soil in the top 75 mm. Forming up the surface soil from the traffic lanes was also important, because root concentrations were six to 20 times smaller in compacted surface soil than in traffic-free soil.

The volume of soil for roots is further increased by better use of the B horizon clay, through loosening the whole solum of the bed area to about 500-mm depth. The C horizon provides drainage of the modified solum. The subsoil to be loosened should be moderately moist (near the plastic limit), achieved by a light irrigation or rain or, if too wet, by drying by the cover crop. The deep tillage and surface soil tillage must produce fragments <20 mm diameter but with no fine fragments (Olsson et al., 1995). The loosened soil must be protected from re-compacting.

2. Reducing mechanical resistance

High mechanical resistance can be either an intrinsic property of the soil, such as in natural pans and hard-setting, or induced, such as by traffic compaction. Deep tillage breaks up the dense surface soil and subsoil. However, the soil rapidly hardens again. Much of the loosened soil consists of fragments that retain the high strength of the original. It is difficult for roots to penetrate dense fragments larger than about 20 mm (Dexter, 1988). In addition, the fine material between the fragments slakes and coalesces to a strong matrix during subsequent wetting and drying cycles under non-tillage. Coalescence is the slow hardening of water-stable soil caused by welding of aggregates to each other, by swelling and overburden pressures, and during movement of fine particles to points of contact of the larger aggregates. These processes must be prevented to achieve the aim of low strength over the whole range of water contents.

Mechanical resistance of loosened surface soils and subsoils <0.5 MPa have been maintained in experiments for up to four years or more (B. Cockroft, unpublished data) by taking into account the processes for forming and stabilizing aggregates, set out in Table 16.2. The main issues are to increase soil Ca and organic matter, maintain root growth, irrigate slowly, control soil water to avoid very low suctions, and prevent compaction by traffic (Tisdall et al., 1984). Gypsum is added to supply Ca and prevent dispersion; lime is added to correct acidity. Slaking can be reduced by maintaining active roots the year round, through the cover crop in winter and the trees in summer. The roots help prevent fine soil particles from coalescing to form a strong matrix by the processes set out in Table 16.2; they also soften, even though slowly, the larger fragments of the original deep tillage (Dexter, 1988). Slow wetting by low-rate spray irrigation reduces slaking and coalescence. Any practice that prevents the soil water suction from coming to zero helps to prevent hardening. The mechanisms involved in producing and managing soft aggregates are discussed by Olsson et al. (1995).

3. Improving soil water

For high infiltration rates the soil must be stable in water and maintain >15% macropores (Table 16.1). The very low macroporosity and low permeability of the clay subsoil, however, restricts water penetration and also impedes drainage. The rate and amount of water flow to roots are limited by the few storage pores and low hydraulic conductivity (Olsson and Rose, 1978; 1988). The typical B horizon of a duplex soil has less than 12% storage pores by volume; its hydraulic conductivity at 40 kPa matric suction is in the order of 10^{-7} m day^{-1} (Olsson et al., 1995). Water flow to the sparse roots is much slowed as the soil dries, so that potential transpiration rates, needed for high yields, are not maintained (Rab and Willatt, 1987; Olsson and Rose, 1988). Shallow wetting further limits the supply of water.

Deep tillage initially improves the water supply through faster infiltration, greater depth of wetting, and higher root concentrations in the B horizon (Taylor and Olsson, 1987). However, deep tillage on its own only partly improves water supply. This is because the fragments formed by the tillage initially have the hydraulic properties of the massive clay layer, that is, they store little water and have a high resistance to water flow. In addition, the fines between the fragments coalesce to a dense matrix with hydraulic properties similar to those of the untreated soil. Passioura (1991) showed that roots have difficulty accessing water stored in the hard fragments at sufficient rates to maintain potential transpiration rates. This soil structural condition can be improved by maintaining permanent root growth and other inputs (Table 16.2) to encourage soil aggregation (Tisdall and Oades, 1979). Olsson et al. (1995) describe increases in unsaturated hydraulic conductivity in a fragmented B horizon in rhizotron chambers. The main factors involved were Ca status and crop roots.

4. Improving soil aeration

Duplex soils do not drain to the required air-filled porosity in 24 hours (Table 16.1) under traditional management, due to the low permeability of the B horizon. The consequent poor aeration can kill trees in wet years and after excessive irrigation (Skene and Poutsma, 1962). Drainage of the solum is improved by better surface drainage, careful control of irrigation, providing flow to the permeable C horizon by deep tillage, and water absorption by cover crops in winter. The deep tillage improves aeration and, if done properly and stabilized, maintains an air-filled porosity of >15% in the bed to 500-mm depth, even though the soil has coalesced. During irrigation, slow application by fine sprinklers minimizes the filling of the larger macropores with water.

5. Reducing soil temperature

In these soils the shallow root systems are susceptible to high temperatures in the surface soil in summer (Table 16.1); under direct sunlight, the top 70 mm can exceed the lethal 35 °C (Cockroft and Tisdall, 1978). Shading by the leaves of the trees will keep temperatures below 25 °C, and the mulch from the dead cover crop will reduce this even further to 22 °C. The loose, aggregated soil in the new system further reduces the heat flux into the surface soil compared to conventional soil.

B. Row crops

The row crops used include maize (*Zea mays* L.), soybean (*Glycine max* L. Merr.), other grain and fodder crops, and most vegetables. The new system develops that of Adem et al. (1982) by using deep and careful tillage, no mulch, roots to reduce coalescence, lime, and low rate irrigation.

The initial soil survey, amendments, and grading are similar to those for orchards. The grower sets up beds, commonly 1.5 m between centres or hills at 0.75 m centres. Crops grow best on beds or hills because the soil drains better than on flat land, the crop roots have a greater volume of uncompacted soil, and there is less soil slaking. Traffic is confined to the furrows and the furrows are used for irrigation. The grower then deep-tills to the required depth down the centre of each bed with one tine or one deep tine following a shallow one. The grower re-forms the beds and subsequently confines all traffic to the furrows between the beds.

The grower prepares the beds or hills in autumn for a summer crop and then sows ryegrass or other crops on the beds as soon as possible. The crop roots then reduce coalescence, hard-setting, and crusting of the soil. In spring the grower kills the grass, chops and cultivates it in, and sows the crop. The crop roots then maintain the stability of the soil structure. The grower uses a system of sprinkler, drip, or furrow irrigation designed to wet the soil slowly, and tries to avoid saturating any part of the profile. Clearly, flood irrigation is avoided. In heavy rain, the beds and slope of the furrows shed water rapidly and help to prevent excessive wetness and structural collapse. The grower can reduce any excessive infiltration into the bed from furrow irrigation by slaking the furrow face or compacting it with wheels. Before each subsequent crop the grower examines the state of the soil profile, assessing soil looseness and recompaction to decide on the need for tillage. Any tillage must be gentle, conducted when the soil is near the plastic limit to avoid powder and clods. So, modern soil management involves minimum tillage, deep tillage, slow wetting, and growing as much root as possible, with the crop grown on beds. Practices such as no treatment of restricting layers, bare cultivated fallow, cultivation during the life of the crop, and flood irrigation are counter-productive to good soil management.

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Chapter 17

CASE STUDIES OF SOIL QUALITY IN THE CANADIAN PRAIRIES: LONG-TERM FIELD EXPERIMENTS

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I. INTRODUCTION

Long-term monitoring is required to assess the effects of natural and anthropogenic processes on soil-quality attributes. Long-term crop-rotation experiments were set up in the Canadian Prairies with the prime goal of improving crop yields and economic returns. These experiments can be used to assess changes in physical, chemical, and biological attributes of soil quality, provided that climatic and edaphic factors are carefully evaluated.

In this chapter, we have compiled data from four sites in western Canada (Fig. 17.1) that have been continually cropped for 28 to 64 years. Three sites are located on soils developed under grassland vegetation (Chernozemic soils). Two of these sites (Swift Current, Sask., and Lethbridge, Alta.) are located in the Brown

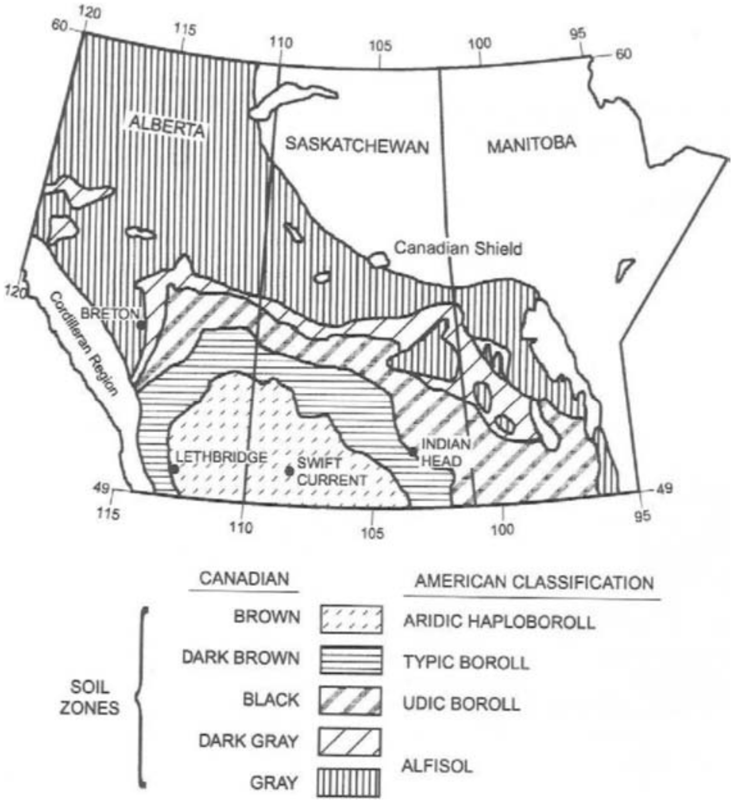

Fig. 17.1. Soil zones of the Canadian prairies and location of the four long-term study sites discussed in this paper.

(Aridic Haploboroll) and Dark Brown (Typic Boroll) soil zones, which generally experience severe water deficit over the growing season (Campbell et al., 1990). The third Chernozemic soil is a thin Black Chernozem (Udic Boroll), which generally does not experience a moisture deficit over the growing season. The fourth site (Breton) is located on a Gray Luvisol (Alfisol) that developed under forest vegetation and rarely experiences moisture deficit during the growing season.

In general, cultivation of Chernozemic soils has resulted in a 15–30% loss of soil organic matter (Campbell, 1978; McGill et al., 1981), and management practices have been developed to reduce the rate of organic matter loss (Campbell et al., 1986). In contrast, the organic matter content of marginal soils developed under forest vegetation has remained constant or increased through cropping and fertilization. Compilation of data from the long-term experiments at our four sites can provide information that can be used to develop indicators of sustainable land use and to assess the impact of management on sustained productivity of these soils. The objective of this work was to document changes in soil quality attributes in the long-term rotations at these four sites and to test and select minimum data sets to assess changes in soil quality. The attributes examined differed somewhat by site.

II. SITE 1: SWIFT CURRENT, SASKATCHEWAN

A. Background

The most comprehensive long-term crop-rotation study conducted in the semi-arid Brown Chernozemic soil zone (Aridic Haploboroll) of the Canadian prairies is an ongoing experiment initiated in 1966 at Swift Current, Sask. (Campbell et al., 1990). The Brown soil zone occupies 5.9 million ha of arable land in a fragile ecosystem. The native vegetation was mainly xerophytic and mesophytic grasses and forbs. Soils in this area were first cultivated at the beginning of the 20th century. These fertile soils require moderate additions of N and P, and water is the main constraint to production. The latter has resulted in widespread use of summerfallow to store extra soil water. However, this practice, which usually involves several cultivations, has encouraged soil erosion, loss of soil organic matter, and increased salinity, thereby leading to a decline in soil fertility and crop productivity (Campbell et al., 1976; McGill et al., 1981).

The various treatments of this rotation experiment allow us to examine the influence of fertilization, legume pulse crops, cropping intensity, and crop type on soil biochemical, chemical, and physical attributes that influence soil and environmental quality, and crop production.

B. Climate and geography

Swift Current is situated at 50°17′N, 107°48′W, in southwestern Saskatchewan. The continental climate prevalent in the region is characterized by short, dry, hot summers and long, cold winters (Campbell et al., 1990). July is the warmest and January the coldest month. The mean annual temperature is 3.5 °C, and the difference in temperature between July and January is 33 °C. Daily temperature differentials of 20–25 °C are not unusual. About 50% of the annual precipitation occurs between 1 May and 30 September, and about 30% comes as snow in winter. Snow is potentially an important source of available water and it insulates and protects the soil from erosion. At Swift Current there is usually a large water deficit in the growing season (Campbell et al., 1990); thus, stored water is very important. June has the highest precipitation and July the highest potential evapotranspiration (ETp). The mean annual precipitation is 358 mm, and ETp is 729 mm, for a mean annual deficit of 371 mm. Furthermore, Swift Current is one of the windiest locations on the Canadian prairies (Campbell et al., 1990). Frost-free days (>0 °C) number 117 on average, among the most on the prairies.

The soil is a Swinton loam to silt loam, an Orthic Brown Chernozem, developed from aeolian deposits overlying glacial till. The organic C and N contents of the 0- to 15-cm depth are 20 and 2.1 g kg^{-1}, respectively, the pH in water paste is 6.75, and the average bulk density is 1.22 Mg m^{-3}. The experimental site is situated on gently sloping land (<3%). The Ap horizon is about 8 cm thick, with moderate, medium subangular, blocky primary structure (Ayres et al., 1985).

C. Description of experiment

Starting in the 1930s, before this experiment began, the land cultivated since the early 1900s was used for various cereal experiments and was managed under a two-year summerfallow–spring wheat (*Triticum aestivum* L.) system. Weeds were mainly controlled by tillage using a cultivator and/or rodweeder. Details of the design and management of this crop rotation experiment have been reported (Campbell et al., 1983b; Biederbeck et al., 1984; Campbell et al., 1992c); thus, only a brief outline of the treatments, management, sampling, and types of analysis is presented here.

The study originally consisted of 12 crop rotations with various sequences of spring wheat, fall rye (*Secale cereale* L.), flax (*Linum usitatissimum* L.), and summerfallow (Table 17.1). All phases of the rotations were represented each year. The rotation was laid out in a randomized complete block design with three replicates. The original design was amended in 1979 by including a rotation of spring wheat with grain lentil (*Lens culinaris medikus* L.), and in 1985 by replacing fall rye (conventional fallow) with winter wheat (chemical fallow). Designated treatments (Table 17.1) received N fertilizer (ammonium nitrate) according to regional recommendations based on soil nitrate concentrations measured in the previous fall. Selected treatments also received monoammonium phosphate placed with the seed at a rate of 10 kg P ha^{-1}.

Soil samples were taken periodically, from various depths, to assess several biochemical, chemical, and physical attributes that can be used to characterize soil quality. These attributes include: soil organic matter, microbial biomass, light fraction organic matter, and microbial activity (C and N mineralization) (Biederbeck et al., 1994), soil pH (Campbell and Zentner, 1984), soil aggregation (Campbell et al., 1993a), and the disposition of available N and P in soil (Campbell et al., 1983a, 1984b; Campbell and Zentner, 1993).

D. Results and discussion

The biochemical and physical characteristics of a soil are, to a great extent, a function of the amount and quality of the crop residues returned to the soil (Rasmussen et al., 1980; Campbell and Zentner, 1993). Thus, it is worth noting how the various crop management systems have influenced crop residue input, which is a function of crop production (Campbell and Zentner, 1993). Yields from these systems have been discussed in detail elsewhere (Zentner and Campbell, 1988; Campbell et al., 1992c; Campbell and Zentner, 1993).

1. Yield trends and grain production

When attempting to determine if there are any trends in crop production over time, it is common practice to calculate a running mean over time, which, it is hoped, will dampen the responses to weather. Thus we calculated five-year running means of yields for wheat grown on stubble (stubble-crop wheat) in the fallow–wheat–wheat (F–W–W) and continuous wheat (Cont W) rotations to determine if there was a degrading influence due to inadequate fertilization or an agrading trend due to

TABLE 17.1

Crop rotations and fertilizer treatments at Swift Current, Sask., showing influence on grain production per rotation per year (1967–1990)

Rotation[a]	Criteria	Average fertilizer application to rotation (1967–1990) N	P	Grain production
		(kg ha^{-1} yr^{-1})		(kg ha^{-1} yr^{-1})
F–(W)	N and P applied as required	3.2	6.4	940
F–W(W)	N and P applied as required	11.2	6.4	1094
(F)–W–(W)	P applied as required but no N applied except that in P fertilizer	3.9	6.4	1057
F–W–W	N applied as required, no P applied	6.2	0	985
F–(Rye)–W[c]	N and P applied as required	10.4	6.0	1078
CF–WW–WW	N and P applied as required	9.4	4.8	
F–Flx–(W)	N and P applied as required	9.5	5.6	705
Cont (W)	N and P applied as required	29.7	9.6	1318
Cont (W)[d]	(Fallow if less than 60 cm moist soil exists at seeding time: N and P applied as required)	27.5	9.6	–
Cont (W)[d]	(Fallow if grassy weeds become a problem: N and P applied as required)	29.2	9.6	–
(W)–Lent[d]	N and P applied as required	20.4	9.1	–
Cont (W)[e]	P applied as required but no N applied except that in P fertilizer	9.0	9.6	1155

[a]Selected plots, indicated in parentheses, were sampled for straw weight and straw N and P concentration at harvest. (F = fallow; CF = chemical fallow; W = spring wheat; WW = winter wheat; Rye = fall rye; Flx = flax; Lent = grain lentil; Cont = continuous).
[b]These values include the fallow year (e.g., for F–W the value is yield of wheat ÷ 2; for F–W–W it is the yield of two years of wheat ÷ 3).
[c]After 1984, this rotation was changed to chemical fallow–winter wheat–winter wheat (spring wheat whenever winter wheat failed to survive the winter).
[d]During the first 12 years, the criteria necessary for summerfallowing in these two rotations were met on several occasions but the rotation change was not implemented. In 1979, these two Cont W rotations were changed to the spring wheat–lentil rotation.
[e]In 1980 and 1982, N was inadvertently applied to this system at rates of 70 and 40 kg N ha^{-1}, respectively.

annual fertilization. The positive influence of fertilizer on yields was apparent in both the fallow and Cont W systems; however, only the latter system, particularly in the last six years of above-average yields, has shown evidence of the soil-agrading influence of annual cropping coupled with proper fertilization (Fig. 17.2). A similar effect was observed in the sub-humid thin Black Chernozem at Indian Head (Campbell et al., 1993c). Zentner et al. (1993b) had shown that yields of stubble wheat (but not wheat grown on fallow) were significantly ($P < 0.05$) correlated with stored soil moisture in the spring, and once the effects of stored soil moisture and growing season precipitation were accounted for, yield trends with time (up to 1992) were not significant. This did not surprise us, because in the semi-arid environment at Swift Current, with its low nutrient export (harvest, leaching, and denitrification), we did not expect to observe any great deviation too early. However, the trend observed for Cont W must have been supported by the six successive years (1989–1995) of above-average precipitation.

The amount of grain produced per rotation each year was calculated by summing the yields for each rotation phase (including no yields for summerfallow) and dividing by the number of rotation phases in the rotation (Table 17.1). For

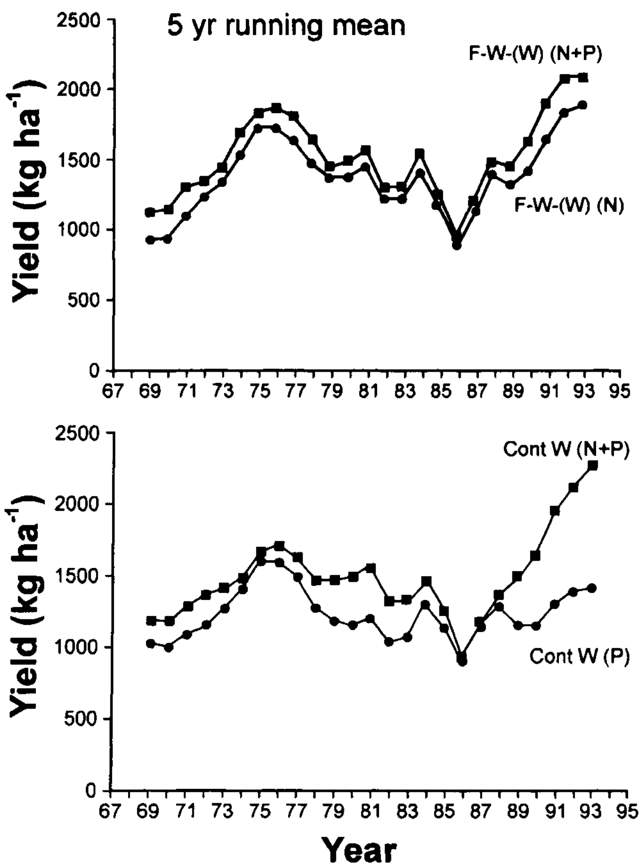

Fig. 17.2. Yield trends of hard red spring wheat grown on stubble at Swift Current, Saskatchewan, showing influence of fertilizer. (The points are five-year running means). In 1988 there was complete crop failure due to drought.

example, for Cont W the grain production would be the same as the unit area yield, while for F–W it would be one-half the unit area yield. On this basis, grain production increased with cropping frequency and with fertilization. These results are similar to those reported for most rotation experiments on the Canadian prairies (Campbell et al., 1990). Substituting a fall-seeded crop for spring wheat in the fertilized three-year rotation had little influence on grain production, but substituting the low-yielding oilseed crop (flax) for spring wheat markedly decreased grain production.

2. Soil biochemical characteristics

The effects of the different treatments on soil biochemical characteristics were assessed after different periods (Biederbeck et al., 1984; Campbell and Zentner, 1993). The most recent analysis of this type was done on samples taken from the 0- to 7.5- and 7.5- to 15-cm depths (Biederbeck et al., 1994), and it is these results that form the basis of the following discussion.

One characteristic measured was the quantity of soil organic matter. Because soil organic matter is chemically and biologically heterogeneous (Campbell, 1978), it is important when assessing the influence of crop management systems on the ability of soils to sustain crop production to examine changes, not only in the amount, but also in the forms of soil N. Although the various labile fractions constitute only a small proportion of the total organic matter in soils, they account for much of the fluctuation in available N over time and are closely related to soil fertility and soil physical properties (Campbell, 1978; Camberdella and Elliott, 1992; Campbell and Zentner, 1993). These labile fractions (e.g., amino compounds, microbial biomass, and light fraction organic matter) and their activity (e.g., C and N mineralization and specific respiratory activity, SRA) are therefore important indices of soil organic matter quality.

Quantity of soil organic matter: Organic N content in the surface soil layer was significantly affected by cropping treatment (Table 17.2). Nitrogen content was positively related to cropping frequency. As well, N fertilization increased soil organic N of the Cont W treatment, but P application had no effect on the F–W–W system. The rotation that included the fall-seeded crop maintained comparatively high soil N content throughout the study period, partly because of reduced tillage intensity (chemical fallow), a shortened fallow period, and the more efficient use of N that is a characteristic of fall-seeded crops (Campbell et al., 1984a; Campbell and Zentner, 1993).

Organic C content followed trends similar to those described for N, as shown by the constant C/N ratio, but differences in organic C between rotations were significant only at $P = 0.09$ (Biederbeck et al., 1994). The organic matter contents were comparable to those reported earlier for this experiment (Biederbeck et al., 1984; Campbell and Zentner 1993).

Carbon and N content of the subsurface soil (7.5 to 15 cm) responded in a manner comparable to that of the surface layer, but values were slightly lower and differences between rotations were not as significant (Table 17.2).

TABLE 17.2

Selected soil biochemical characteristics of a Brown Chernozemic soil at Swift Current, Sask., as influenced by crop rotation and fertilization. Soils were sampled on 5 October, 1990, 24 years after the rotations were established (from Biederbeck et al., 1994)

Rotation phase sampled	Specific Fertilizer[a]	Organic matter		Microbial biomass[b]		Light fraction		Cumulative mineralization[c]		Respiratory activity[d]
		C	N	C	N	C	N	C	N	
0–7.5 cm		(t ha^{-1})		(kg ha^{-1})		(t ha^{-1})	(t ha^{-1})	(kg ha^{-1})		
(F)–W	N,P	14.8	1.47	222	27	1.01	0.05	136	69	0.61
(F)–W–W	N,P	16.3	1.73	221	28	1.34	0.08	159	83	0.74
(F)–W–W	N	16.0	1.73	213	31	1.33	0.08	121	74	0.60
(F)–Flx–W	N,P	15.4	1.64	242	40	1.71	0.10	182	87	0.75
(CF)–WW–WW	N,P	17.7	1.90	226	45	1.85	0.12	177	99	0.78
(W)–Lent	N,P	17.5	1.73	267	41	1.90	0.12	239	87	0.94
Cont(W)	N,P	18.4	1.90	317	45	2.72	0.16	320	109	1.02
Cont(W)	P	16.4	1.64	340	58	1.69	0.10	302	87	0.89
Signif.(P)[e]		0.09	0.01	0.01	0.01	0.002	0.002	0.001	0.05	0.05
LSD($P < 0.05$)		2.3	0.17	74	16	0.61	0.03	43	23	0.27
CV(%)		8	6	16	23	21	18	12	15	20

TABLE 17.2 (continued)

Rotation phase sampled	Specific Fertilizer[a]	Organic matter		Microbial biomass[b]		Light fraction		Cumulative mineralization[c]		Respiratory activity[d]
		C	N	C	N	C	N	C	N	
7.5–15 cm										
(F)–W	N,P	15.5	1.65			0.43	0.02	41	52	
(F)–W–W	N,P	17.8	1.94			0.68	0.04	51	63	
(F)–W–W	N	16.8	1.84			0.74	0.04	63	77	
(F)–Flx–W	N,P	16.1	1.74			0.72	0.04	57	71	
(CF)–WW–WW	N,P	17.9	1.94			0.71	0.04	59	84	
(W)–Lent	N,P	17.3	1.74			0.99	0.06	164	83	
Cont(W)	N,P	18.0	1.94			1.07	0.07	151	88	
Cont(W)	P	15.7	1.65			0.78	0.04	131	72	
Signif (P)[e]		NS	0.09			NS	NS	0.001	0.003	
LSD(P < 0.05)		–	0.19			–	–	28	15	
CV(%)		10	7			38	37	18	12	

[a]N and P denote application of nitrogen and phosphorus as shown in Table 17.1.
[b]Microbial biomass was only determined in the 0- to 7.5-cm depth.
[c]C mineralization determined by incubating field moist soil wetted to field capacity at 21 °C for 30 days; N mineralization determined by incubating re-wetted air-dry soils at 35 °C for 16 wk with intermittent leaching.
[d]Specific respiratory activity = Cum. C mineralization/microbial biomass C.
[e]NS = not significant at P = 0.10.

Amino compounds in soil: Biederbeck et al. (1986) sampled the 0- to 7.5-cm depth of selected rotations at Swift Current in spring 1984 and fractionated these soils by acid hydrolysis to determine amino constituents (Bremner, 1965; Stevenson, 1982). They found that the total amount of amino acid N was directly proportional to cropping frequency and was increased by N fertilizer in the Cont W system (Table 17.3). However, when the relative molar distribution of amino acids was used as an index of changes in soil organic N quality, no differences were found between the rotations. These results suggest that harsh acid extractions are not sensitive in delineating changes in soil organic matter quality, although they may be able to identify treatments that contribute large quantities of N to the system.

Soil microbial biomass: Soil microbial biomass C and N in the surface 7.5 cm of this soil was significantly suppressed by the inclusion of summerfallow in the rotation (Table 17.2). This observation corroborates those of other studies (Schnürer et al., 1985; Carter, 1986; Campbell et al., 1991a; Collins et al., 1992) and is largely attributable to variation in amount of residue input and timing of incorporation. Soil from Cont W had higher biomass C and N content than that from the wheat–lentil (W–Lent) treatment, presumably because of differences in substrate composition.

Microbial biomass C in surface soil from the Cont W rotation was virtually identical to that reported in earlier sampling (Biederbeck et al., 1984, 1986; Biederbeck and Campbell, 1987). The biomass C values in the rotations with summerfallow were slightly lower than those reported in earlier studies, in which biomass was measured in the cropped phase of the rotation, but still showed similar responses to cropping systems. These findings suggest that microbial biomass is reasonably stable and consistent over time, despite minor variations with rotation phase. The estimates of biomass in this study were lower than those reported for similar soils in more humid environments (Carter, 1986; Campbell et al., 1991a, 1992a), probably because of differences in the amount and quality of residue input and constraints to microbial activity in the more arid soil environment at Swift Current.

TABLE 17.3

Kjeldahl-N and N compounds extracted in hydrolysis of soil in 0- to 7.5-cm depth with 6*M* HCl for selected crop rotations at Swift Current, Sask., 1984 (adapted from Biederbeck et al., 1986)

Rotation	Fertilizer	Type of N Compound			Total N
		Amino acid-N	Amino sugar-N	Ammonia-N	
			— (kg ha^{-1}) —		
F–(W)	N,P	563	75	505	2208
F–W–(W)	N,P	653	85	537	2576
F–W–(W)	N	635	75	529	2484
Cont (W)	N,P	700	92	560	2576
Cont (W)	P	653	87	558	2484

Withholding N fertilizer tended to increase microbial biomass content of Cont W (Table 17.2). Although this response was not statistically significant, it has been consistently observed in all of the previous analyses at this site and in similar soils elsewhere (Biederbeck et al., 1984, 1986; Biederbeck and Campbell, 1987; Collins et al., 1992). Biederbeck et al. (1984) speculated that severe N deficiency resulted in a microbial population with a comparatively high proportion of dormant cells.

The C:N ratio of the microbial biomass, which ranged from 5.0 to 8.3, was unaffected by cropping treatment (Table 17.2). Similarly, the proportion of the total organic C or N present in microbial biomass was only marginally affected by cropping treatment (Biederbeck et al., 1994). Thus, the microbial C/organic C ratio, which has been proposed by Sparling (1992) as an indicator of changes in soil organic matter quality, may not be particularly sensitive within a particular climatic region such as the semi-arid conditions found in southwestern Saskatchewan.

Light fraction organic matter: The amount of light fraction organic matter (LF-OM; separated using NaI solution with specific gravity of 1.7 g cm^{-3}) in the surface soil layer was highly responsive to cropping treatment (Table 17.2), accounting for between 7 and 15% of the total organic C and between 3 and 8% of the total organic N in this soil (Biederbeck et al., 1994). As observed by Janzen et al. (1992), these differences in LF-C accounted for a large proportion of the variation in total organic C ($r^2 = 0.66$, $P < 0.01$). The LF-OM, whether expressed on the basis of dry matter (DM), C, or N content, was inversely related to fallow frequency. For example, the LF-C content of the surface soil of the Cont W (N + P) treatment was almost three times that of the F–W treatment (Table 17.2). Nitrogen fertilizer significantly enhanced soil LF in Cont W. Inclusion of winter wheat in the three-year rotation (along with the use of chemical fallow) tended to increase the LF content over that observed in the conventional summerfallow–spring wheat rotations. The W–Lent system had LF contents intermediate between those of well-fertilized Cont W and F-W. Replacing spring wheat on summerfallow with flax in the F–W–W rotation did not affect LF content.

Values in the 7.5- to 15-cm soil layer were generally less than half of those in the surface layer (Table 17.2). Responses in the subsurface layer were similar to those in the surface layer, but differences were not significant.

Mineralizable C and N and specific respiratory activity: As noted by Campbell et al. (1991a; 1992a), C mineralization was sensitive to differences between treatments (Table 17.2). As observed for LF, C mineralization in the surface layer was negatively related to frequency of fallow. Among the continuously cropped treatments, C mineralization was higher in soil under monoculture wheat than in that of the W–Lent rotation. Withholding fertilizer tended to depress C mineralization, particularly in the F–W–W rotation. The proportion of total organic C that was mineralized varied among treatments (Biederbeck et al., 1994), suggesting that differences in C mineralization were at least partly attributable to variation in OM quality.

Carbon mineralization in the 7.5- to 15-cm layer was also affected by treatments, especially fallow frequency (Table 17.2). Responses in this layer paralleled those described in the surface layer, but the proportion of organic C mineralized was lower (Biederbeck et al., 1994), indicating that subsurface OM was more resistant to decomposition.

Nitrogen mineralization responses generally paralleled C mineralization, but differences in N mineralization among treatments were smaller (Table 17.2). The withholding of N fertilizer in Cont W significantly depressed N mineralization. In contrast to observations for C, the proportion of the total organic N mineralized in the subsurface layer (mean = 4.1%) was not appreciably lower than the proportion mineralized in the surface layer (mean = 5.1%).

The specific respiratory activity (C mineralization/biomass C) has been proposed as a measure of substrate availability in surface soils (Schnürer et al., 1985; Anderson and Domsch, 1990). This ratio was highest in continuously cropped treatments and lowest in fallow–wheat rotations (Table 17.2). Withholding N in the Cont W treatment tended to reduce the specific respiratory activity, supporting the hypothesis that there was an increase in dormant cells in this treatment (Biederbeck et al., 1984). These responses, similar to those reported for a Dark Brown Chernozem (Campbell et al., 1992a), support the use of specific respiratory activity as one index of substrate availability in semi-arid soil environments.

Factors controlling labile OM: Perhaps the best estimate of labile OM is that provided by C and N mineralization, because these characteristics give a direct measure of OM turnover. Light fraction C was significantly correlated to C mineralization (r = 0.68 and 0.60, $P < 0.05$ in surface and subsurface soil, respectively), and light fraction N was significantly related to N mineralization (r = 0.50 and 0.56, $P < 0.05$, in the surface and subsurface soil, respectively).

Microbial biomass C (MB-C) was the variable most closely correlated with C mineralization (r = 0.80, $P < 0.01$, in surface soil). This confirms biomass C as a reliable indicator of labile OM, as reported by others (Carter, 1986; Campbell et al., 1991a; Angers et al., 1993). This close relationship is to be expected because microbial biomass is substrate and the microbes are agents of soil respiration.

When both microbial biomass C and LF-C in the 0- to 7.5-cm depth were included in a multiple regression, they accounted for almost 98% of the variability in C mineralization (n = 8) (Biederbeck et al., 1994):

$$C_{min} = -176 + 1.19 \text{ MB-C} + 0.044 \text{ LF-C} \quad (1)$$

where C_{min} = C mineralization in 30 days at 21 °C. (All values expressed as kg ha^{-1}). The standardized estimate of the parameters were 0.77 for MB-C and 0.30 for LF-C, indicating that the influence of MB-C on C mineralization was 2.6 times that of LF-C. This equation suggests that both independent variables are important, yet distinct, substrates for C mineralization.

The amount of labile OM in soil of a given treatment is a function of the relative rates of residue input and decomposition. Biederbeck et al. (1994) estimated the recent and long-term inputs of residue C from the grain yields (Table 17.4). Residue C was higher in continuously cropped treatments than in those with summerfallow. Further, N fertilization of Cont W tended to enhance C inputs, but

TABLE 17.4

Estimated inputs of C from crop residues as influenced by crop rotation and fertilizer application (from Biederbeck et al., 1994)

Rotation	Fertilizer	1990	1989	1988	1985–90[b]	1979–90[b]	Residual C[b]
		Estimated C inputs (Mg ha^{-1})[a]					
Cont (W)	N,P	1.72	1.66	0	7.79	16.2	3.37
Cont (W)	P	1.33	1.70	0	7.24	13.8	2.95
(W)–Lent	N,P	1.48	2.02	0	8.22	14.9	3.36
(CF)–WW–WW	N,P	0	1.35	0.89	5.40	12.1	1.47
(F)–W–W	N,P	0	1.59	0.89	5.91	13.5	1.63
(F)–W–W	N	0	1.50	0.71	5.47	12.6	1.51
(F)–Flx–W	N,P	0	1.66	0.76	5.69	10.1	1.60
(F)–W	N,P	0	2.12	0	5.03	11.8	1.56
Significance (P)[c]		NS	0.0003	NS	0.0001	0.0001	0.0001
LSD (P = 0.05)			0.25	–	0.74	1.3	0.37

[a]Straw yields were estimated as follows: for wheat, straw yield = 1.5 × grain yield (Campbell et al., 1977); for flax, straw yield = 696 + 1.17 × grain yield (based on regression analysis of yields from 1985–1992); for lentil, straw yield = 370 + 1.47 × grain yield (based on regression analysis of data from Campbell et al., 1992a). For all crops, root/straw ratio was assumed to be 0.59 (Campbell et al., 1977) and C concentration of tissues was assumed to be 45% (Millar et al., 1936). Yields from plots other than those sampled in 1990 (i.e., other rotation phases) were not included in estimates presented.

[b]'1985–90' and '1979–90' denote cumulative residue C for the last six- and 12-yr period, respectively. These periods were selected to avoid a confounding effect of rotation (six is the lowest common multiple of all rotation lengths); residual C = cumulative residue C remaining from previous six years as estimated from: $y = 0.72e^{-1.4t} + 0.28^{e-0.081t}$, where y = proportion remaining and t = years since residue application (from Voroney et al., 1989).

[c]Significance of one-way analysis of variance, excluding all treatments where estimated C inputs were 0.

P fertilization of the F–W–W system had no effect. Most of the indices of labile OM were positively correlated to the estimated recent (1990) inputs of residues (e.g., for C_{min}, $r = 0.85$, $P < 0.05$; for LF-C, $r = 0.64$, $P < 0.05$; for MB-C, $r = 0.64$, $P < 0.05$; for SRA, $r = 0.69$, $P < 0.05$; and for N_{min}, $r = 0.40$, $P < 0.10$). The positive effect of N fertilization on labile organic matter was probably through its influence on the production of crop residues. Most of the indices were also correlated to residue inputs over the previous six- and 12-year periods (Biederbeck et al., 1994).

Based on the foregoing results, Biederbeck et al. (1994) proposed a procedure for making a first estimate of the sensitivity of the various biochemical attributes to agronomic variables. They calculated the ratio of values obtained for Cont W (N + P)/F–W (N + P) (i.e., treatments that generally gave maximum and minimum values, respectively). For surface soil the ranking was: LF-C ≈ LF-N > C_{min} > N_{min} > MB-C ≈ MB-N > soil C ≈ soil N.

3. Soil chemical and physical characteristics

Over the years, the influence of the various treatments on soil pH, the disposition of available N and P in this soil, and soil aggregation, have been assessed. In this section we discuss some of our main findings regarding these soil characteristics.

Soil pH: Soil acidity is an important soil quality characteristic, affecting the well-being of both plants and the soil microflora. In the Swift Current study, Campbell and Zentner (1984) reported an average pH decline in the surface 15 cm of 0.5 units as a result of 17 years of ammonium nitrate application (supplying N at an average of 35 kg ha^{-1} yr^{-1}) to the Cont W system. There was no effect on pH at lower depths. Most prairie soils are well buffered, and N rates used by farmers are usually modest, thus there is little likelihood that this phenomenon will present serious consequences in prairie agriculture.

Available P: Analysis of the top 15 cm of soil showed that, when no P was applied, the bicarbonate–soluble Pi (Olsen–P) remained constant over the 24 years despite the export of P in grain (Selles et al., 1995). When P fertilizer was applied regularly there was a gradual build-up in Olsen–P at an average rate of 1.0 to 1.7 kg ha^{-1} yr^{-1}. The residual fertilizer P in the soil accumulated as sorbed P and microbial P rather than as Ca–phosphate precipitates (Selles et al., 1995). Although the P fertilized soils showed a marked increase in available P, cereal crops still responded positively to small amounts of seed-placed P (Zentner et al., 1993a). The authors related this response to the need for plants to have a ready source of available P near the seedlings especially under the cool, moist conditions frequently experienced on the Canadian prairies.

Available N: Monitoring of NO_3-N throughout the growing season over the first 12 years of the experiment showed the expected gradual accumulation of NO_3-N in summerfallow soils and drawdown of NO_3-N in cropped soils (Campbell et al., 1983a). Deep-core soil sampling of some of the rotations (Campbell et al., 1984a; Campbell and Zentner, 1993; Campbell et al., 1992c) showed that NO_3 was being leached beyond the root zone in periods of above-average precipitation. This leaching was greatest under systems that included frequent summerfallow, except when winter annual cereals (e.g., fall rye) were grown. Winter cereals make efficient use of N, reduce erosion, and consequently maintain soil organic matter at higher levels than do spring-seeded cereals (Campbell et al., 1984a; Campbell and Zentner, 1993). Systems that were adequately fertilized allowed less NO_3 leaching than those that were poorly fertilized, because greater growth and water and N uptake left less NO_3 in the soil to be leached (Campbell et al., 1984a; Campbell et al., 1993e). Comparison of NO_3 leached under the fertilized W–Lent and the fertilized Cont W systems suggested that closer synchrony between N availability and N uptake under the lentil-containing rotation resulted in less leaching (Campbell et al., 1992c).

To determine how the various cropping systems in this study influence total soil organic N, an apparent N deficit, defined as N exported in grain minus N applied as fertilizer, was calculated for each rotation (Campbell and Zentner, 1993). A regression of soil organic N versus apparent N deficit showed that soil organic

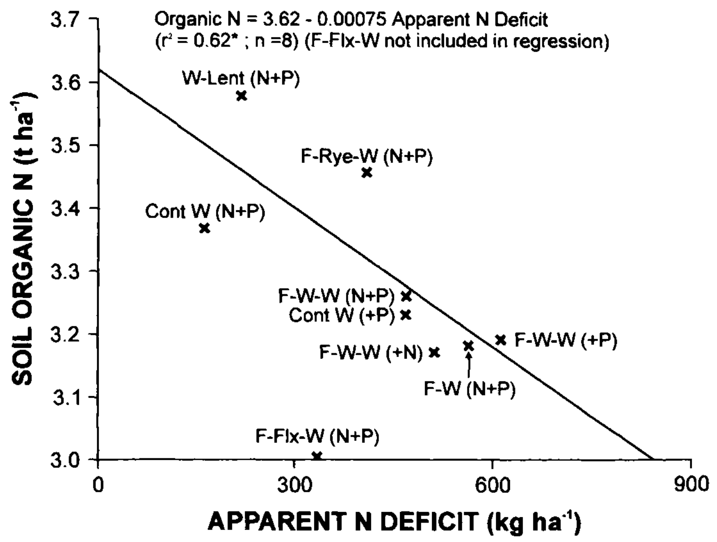

Fig. 17.3. Relationship between soil organic N in the surface 0.15 m of soil, sampled in fall 1990, and apparent N deficit (i.e., N exported in grain minus N applied as fertilizer) (from Campbell and Zentner, 1993). Reprinted with permission from the Soil Science Society of America Journal.

matter decreased as N deficit increased (Fig. 17.3). This emphasizes the need to replace the harvested N by adding fertilizer or growing legumes.

Although the potential N mineralization results (Table 17.2) did not reveal an advantage favouring the W–Lent system over the Cont W (N + P), NO_3-N soil test measurements made in these plots each fall showed that after about five years the grain legume had reduced the fertilizer N requirements of wheat compared to that required by the Cont W (Fig. 17.4). This indicates that, contrary to common belief,

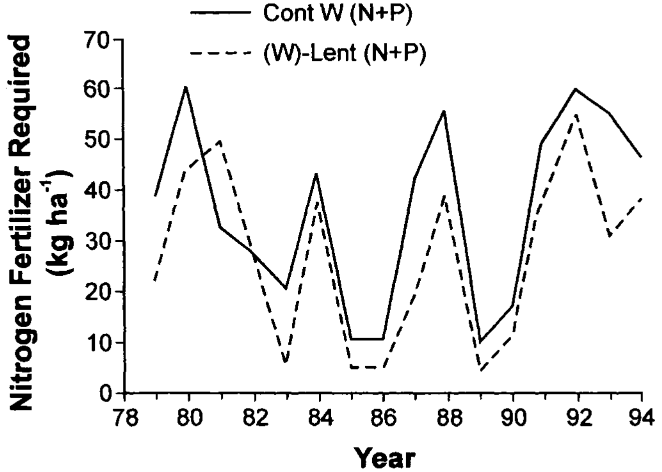

Fig. 17.4. Reduced N fertilizer requirements of cereals grown in rotation with lentil pulse crop at Swift Current, Sask. (adapted from Schoneau and Campbell, 1996).

pulse crops do provide a significant N benefit to soil fertility despite removal of much of the fixed N in the harvested grain (Campbell et al., 1992c). Thus, producers using such systems may reap economic as well as soil quality benefits in due course.

Soil aggregation: The size distribution and stability of aggregates are important characteristics influencing soil tilth, erosion, water infiltration, and nutrient dynamics in soil. Soil management influences soil aggregation.

Analysis of soil samples taken three times during the 1991 growing season from the top 5 cm of the soil in the various crop rotations showed that aggregate size, determined by dry sieving, was directly related to previous precipitation conditions (Campbell et al., 1993a). The presence of crops and the use of management practices that increased crop residues (e.g., fertilizers) reduced the wind-erodible fraction of soil and increased the geometric mean weight diameter of aggregates (Table 17.5). Drought conditions increased water-stable aggregates. Water-stable aggregates were increased by frequent cropping and by adequate fertilization (presumably due to increased production of crop residues) (Table 17.5). Chemical fallow increased aggregate stability, probably as a result of reduced soil disturbance.

TABLE 17.5

Effect of crop rotation and fertilization on geometric mean diameter, GMD[a], geometric standard deviation (GSD) and water stable aggregates[b] at Swift Current, Sask., on two dates in 1991 (from Campbell et al., 1993a)

Crop rotation	Fertilizer	GMD (GSD) (mm)		Water-stable aggregates (%) (fast-wetting[c])	
		3 June	17 Sept.	3 June	17 Sept.
(F)-W	N,P	14.1 (0.03)	4.7 (0.25)	16.2	17.2
F-(W)	N,P	7.2 (0.09)	5.7 (0.17)	15.4	15.1
(F)-W-W	N,P	22.7 (0.11)	6.3 (0.09)	21.8	22.6
F-(W)-W	N,P	6.6 (0.10)	5.7 (0.16)	17.5	22.4
(F)-W-W	N	16.1 (0.07)	4.3 (0.31)	20.4	21.7
F-(W)-W	N	7.0 (0.10)	3.4 (0.43)	23.3	26.9
(F)-Flx-W	N,P	21.1 (0.08)	4.7 (0.23)	20.3	20.4
F-(Flx)-W	N,P	22.1 (0.16)	5.3 (0.13)	14.6	21.2
(CF)-WW-WW	N,P	28.9 (0.15)	7.1 (0.01)	31.8	39.1
CF-(WW)-WW	N,P	5.9 (0.07)	3.1 (0.46)	37.1	41.8
(W)-Len	N,P	8.9 (0.03)	5.4 (0.13)	24.2	34.2
W-(Len)	N,P	13.8 (0.13)	4.8 (0.23)	22.7	31.6
Cont(W)	N,P	8.0 (0.02)	11.1 (0.05)	25.2	40.0
Cont(W)	P	11.9 (0.06)	5.6 (0.08)	25.6	31.3
LSD ($P < 0.05$)[d]					
(Time × Rotation phase): rotation phase = 6, time = 5					

[a]Calculated according to Gardner (1956). All r^2 highly significant. No LSDs calculated for GMDs.
[b]Soil from fraction C of dry sieving (i.e., 1–2 mm aggregates) was used for wet-sieving analyses.
[c]The aggregates were immersed directly in water at start of the wet-sieving procedure.
[d]LSDs calculated for split plot with time of sampling as main plot and rotation phase as subplot.

III. SITE 2: LETHBRIDGE, ALBERTA

A. Background

Several long-term crop rotation studies have been conducted in the Dark Brown soil zone (Typic Boroll) of the Canadian prairies (Campbell et al., 1992a; Bremer et al., 1994; Janzen, 1995). Soils in this zone, developed under short- and tall-grass species, occupy about 11 million ha of land. Most of these soils have been cultivated since the early 1900s.

We will discuss changes in soil quality in the Dark Brown soil zone using, for purposes of illustration, findings from Rotation 116. This long-term site was established in 1951 at the Agriculture and Agri-Food Canada Research Centre in Lethbridge, Alta.

B. Climate and geography

Lethbridge is situated in south-west Alberta, Canada (49°42′N, 112°50′W), approximately 110 km east of the Rocky Mountains. The region is characterized by a continental climate with short, warm summers and long, cold winters. A prominent feature of the climate is the warm chinook (or foehn) winds that moderate the winter air temperature, but also increase evaporation and cause soil erosion.

The mean annual air temperature at Lethbridge is 5 °C, and the frost-free period averages 118 days. On average, the test site has 1689 cumulative growing degree days (>5 °C). Average annual precipitation is 402 mm, and potential evapotranspiration is 681 mm, resulting in a moisture deficit of 279 mm. Snow represents 30% of the annual precipitation, but frequent chinooks minimize the persistence of snow cover and its benefit for moisture replenishment.

Soils in the Lethbridge region were developed under native vegetation of tall and short grasses. At the Agriculture and Agri-Food Canada Research Centre, the soil is a Lethbridge loam to clay loam, an Orthic Dark Brown Chernozem (Typic Haploboroll) developed on alluvial lacustrine parent material. The organic C and N contents of the 0- to 15-cm depth are 16.8 and 1.72 g kg^{-1}, respectively; the pH in water paste is 7.6, and the average bulk density is 1.20 Mg m^{-3}. Typically the soil has a 10- to 15-cm A_p, a 0- to 12-cm B, and a calcareous C horizon. The land was first broken from sod in the early 1900s.

C. Description of experiment

Rotation 116 was established in 1951 on land that had been devoted to mixed crop rotations with "light" manure applications every sixth year since it was broken (Pittman, 1977). Some of the treatments in the experiment have been altered over time. Currently, the experiment includes 13 rotations (we discuss nine in this chapter). Five of these systems have been unaltered (Table 17.6). Most of the other treatments were added in 1985 by random assignment to plots from obsolete treatments, or on adjacent land that had been in a fallow–wheat cropping system. The treatments were arranged in a randomized complete block design with four

TABLE 17.6

Selected crop rotations, nutritive amendments, and grain production in Rotation 116 at Lethbridge, Alta. Production values are means for the entire rotation including the fallow year, except for F–W–W–H–H–H, where the forage phases are excluded

Rotation	Year established	Nutritive amendments[a]	Grain[b] production ($kg\ ha^{-1}\ yr^{-1}$)
Cont W	1951	22 kg P ha^{-1} since 1985	1233
F–W–W	1951	22 kg P ha^{-1} since 1985	1043
F–W	1951	22 kg P ha^{-1} since 1985	906
F_m–W–W[c]	1951	11.2 Mg manure ha^{-1}	1131
F–W–W–H–H–H[d]	1951	22 kg P ha^{-1} since 1985	815
F–W(N)	1985	80 kg N ha^{-1}, 22 kg P ha^{-1}	910
F–W–W–(N)	1985	80 kg N ha^{-1}, 22 kg P ha^{-1}	1107
Cont W (N)	1985	80 kg N ha^{-1}, 22 kg P ha^{-1}	1277
G[e]	1985		

[a]Fertilizer applied prior to seeding of wheat and hay.
[b]From Bremer et al., 1994. Grain yield usually reported at harvest-moisture content, except in a few cases (e.g., 1992) when dry-matter yields were used.
[c]F_m = fallow with animal manure (application rate expressed on basis of wet weight). The annual manure application was estimated to contain 67 kg N ha^{-1}.
[d]Hay was a mixture of alfalfa (*Medicago sativa* L.) and crested wheatgrass (*Agropyron cristatum* L.).
[e]Mixture of *Agropyron* spp.

replicates. Each phase of the 13 rotations is represented every year on plots that measure 3.2 m by 37 m (for a total of 116 plots).

The plots were managed using agronomic practices typical of those recommended to producers in the region. Weeds in cropped phases were controlled using recommended herbicides. In the fallow phase, weeds were controlled using shallow tillage with a wide-blade cultivator, which conserves surface residues. Recently, tillage operations on fallow have occasionally been replaced with an application of a non-selective herbicide. Nitrogen fertilizer (ammonium nitrate) was broadcast on the soil surface prior to seeding, but P (triple superphosphate) was applied with the seed. The native grass plot was not harvested, but the accumulated residues were burned in the spring of 1993 to simulate natural fire cycles on the prairies.

Grain yields were determined by sampling a representative portion of the plot with a small combine. Hay yields were similarly determined by mechanical or manual harvest of a portion of each plot. Nitrogen concentrations in the grain, straw, and hay were determined in recent years (Bremer et al., 1994).

Soil from the various treatments was sampled in 1954 and at regular intervals thereafter. The most recent extensive sampling was performed in September 1992, when soil from the 0- to 7.5-cm, 7.5- to 15-cm, and 15- to 30-cm depths were sampled (Bremer et al., 1994). Analyses performed on these samples included organic C and

N, light fraction organic C and N, and mineralizable C and N (after incubating at 25 °C for 10 wk).

D. Results and discussion

1. Yields

Prior to 1985, grain production per rotation per year was only marginally affected by fallow frequency; production in F–W was slightly lower than in F–W–W, and Cont W was slightly lower than F–W–W in these unfertilized systems (Table 17.6). Inclusion of alfalfa/grass forage in the rotation increased wheat production compared to that in the monoculture wheat rotations, probably because of the additional N fertility derived from symbiotic N_2 fixation (Johnston et al., 1995).

From 1985 to 1992, grain production for the various rotations was comparable to that in the earlier years, except that grain production in Cont W was the highest of the three systems (Table 17.6), possibly responding to the P fertilizer that was applied since 1985. Application of N fertilizer had no appreciable effect on grain production, apparently because yields in these years were constrained by moisture deficiency. Over this period, growing season moisture deficits averaged 613 mm, 44% higher than the long-term average. Wheat yields in the F–W–W–H–H–H rotation were also depressed by water stress (Johnston et al., 1995). Application of farmyard manure (F_m–W–W) increased grain production significantly over those in the other F–W–W treatments (Table 17.6).

2. Soil biochemical characteristics

The soil biochemical characteristics assessed in this study were the amounts of soil organic C and N, light fraction organic matter, and the mineralizable C and N.

Quantity of soil organic matter: Soil organic C content (expressed on the basis of mass per unit area to a depth of 30 cm) was higher in Cont W than in F–W or F–W–W, though differences were small (data not shown). In the treatments to which N was applied since 1985, organic C showed a slight increase due to cropping frequency.

Application of N fertilizer since 1985 has had no effect on organic N, in contrast to findings from related studies (e.g., Campbell et al., 1991a). This result may be partly attributable to the relatively short duration of fertilizer application, and to the comparatively high moisture stress during that time period, which minimized yield response.

Application of manure to the three-year rotation increased organic C content, partly because of the C inputs in the manure and partly due to the higher crop residue inputs resulting from yield response to manure. Soil organic N was also enhanced by manure, although the increase over organic N in the unmanured F–W–W rotation was less than the estimated inputs from manure since 1951 (Bremer et al., 1994).

The highest C and N contents were observed in treatments that contained hay (F–W–W–H–H–H) or native grass (G), even though the latter treatment had been in place for less than eight years. Among the depth increments sampled, the largest differences were observed in the 0- to 7.5-cm layer (Table 17.7). Organic C concentration in this layer ranged from 15.9 g C kg^{-1} in F–W to 21.7 g C kg^{-1} in F_m–W–W. There was no significant difference in organic C among phases of the rotations, indicating that changes in organic matter occurred over a period longer than the duration of these rotations.

Light fraction organic matter: As reported elsewhere (Janzen et al., 1992; Biederbeck et al., 1994), the concentration of light fraction C (LF-C) was more sensitive to management practices than was total organic C (Table 17.7). For example, the LF-C concentration in Cont W was about twice that in F–W, whereas total organic C in Cont W was only 1.2 times that in F–W. Highest concentration of LF-C was

TABLE 17.7

Effect of crop rotation and amendment on total, light fraction, and mineralizable C and N concentrations in surface soil (0–7.5 cm) from Rotation 116 at Lethbridge, Alta. (from Bremer et al., 1994)

Treatment[a]	Organic matter		Light fraction		Cumulative mineralization[b]	
	C	N	C	N	C	N
	— (g kg^{-1}) —		— (mg kg^{-1}) —		— (mg kg^{-1}) —	
F–W	15.9	1.58	1.60	0.10	622	33
F–W–W	17.0	1.63	2.15	0.14	795	34
Cont W	18.7	1.81	3.27	0.23	939	52
F–W (N)	16.4	1.61	1.77	0.12	726	32
F–W–W (N)	16.6	1.60	1.92	0.13	739	40
Cont W (N)	18.5	1.76	2.67	0.19	956	55
F_m–W–W	21.7	2.06	3.56	0.27	983	35
F–W–W–H–H–H	18.6	1.81	2.78	0.20	984	46
G	19.9	1.84	4.75	0.31	1451	43
CV (%)	8	4	27	31	19	27
Contrasts[c]						
Phase	ns	ns	ns	ns	**	ns
Fertilizer	ns	ns	ns	ns	ns	ns
Fallow frequency:						
Cont W vs F–W/F–W–W	*	**	*	**	**	**
F–W vs F–W–W	ns	ns	+	*	*	ns
Manure	*	**	**	**	**	ns
Forage	*	**	**	**	**	**
Grass	ns	ns	**	*	**	ns

[a]For treatment descriptions, see Table 17.1; values in table are mean for all phases of each rotation.
[b]Incubated at 25 °C for 10 wk.
[c]Treatments compared for fertilizer = +N vs −N treatments, manure = F_m–W–W vs. F–W–W ± N, forage = F–W–W–H–H–H vs. F–W–W ± N, grass = G vs. Cont W ± N, ns = not significant (P > 0.10), + = P ≤ 0.10, * = P ≤ 0.05, ** = P ≤ 0.01.

observed in the native grass treatment, where the concentration was three times that in F–W. Application of manure to the F_m–W–W rotation, or inclusion of three years of hay (F–W–W–H–H–H) increased LF-C, though concentrations were still below those in Cont W.

Application of N fertilizer did not affect LF-C in this study. As well, there were no differences in LF-C concentration among phases within rotations, suggesting that the turnover time of the LF material was longer than the duration of the rotations.

The concentration of LF-N followed trends similar to those described for LF-C (Table 17.7), although some significant differences in the LF C/N ratio were observed among treatments (data not shown). The highest LF C/N ratio was observed in the grass treatment (15.7), and lowest ratios were observed in F_m–W–W (13.6), Cont W(+N) (14.0), and F–W–W–H–H–H (14.0), probably because of N inputs in the latter treatments.

Mineralizable C and N: Mineralizable C, as measured from the evolution of CO_2 in a 10-wk laboratory incubation, was closely related to LF-C (Table 17.7). Mineralizable C concentrations increased as frequency of fallow was reduced, and was highest in the grass treatment. Application of N fertilizer did not affect mineralizable C, whereas application of manure and inclusion of forage in rotation increased mineralizable C concentration.

Mineralizable N concentration was less responsive to management treatments than was mineralizable C. For example, the treatment showing the highest mineralizable C (grass) showed lower N mineralization than continuous wheat. Differences in the patterns of C and N mineralization are probably attributable to effects of N immobilization (Janzen et al., 1992) and variation in C/N ratio of organic substrates.

Reasons for differences in soil quality: The variation in organic matter content among treatments was partly attributable to differences in C inputs. Thus, for example, the higher C content in the manured treatment (F_m–W–W) can be attributed partly to enhanced input from higher yields and partly to direct inputs of C in the manure. However, the differences in soil organic matter content, particularly in the surface soil layer, cannot be attributed solely to variable residue inputs. For example, although grain yields (and hence residues) among F–W, F–W–W, and Cont W did not differ greatly during the first 30 years of the study, surface soil C concentration in Cont W was 18% greater than that in F–W. Differences in labile organic matter fractions were even larger.

We postulate that differences in organic matter content and quality among treatments are partly attributable to variable rates of residue decomposition. Because soil moisture conditions are more favourable for decomposition in treatments with frequent fallow, labile C substrates do not accumulate. In continuous cropping treatments, or under grass, decomposition may be constrained by moisture deficiency because plant growth removes available water from the soil; thus decomposable C will accumulate to a greater degree than in fallow systems. The labile fraction of soil organic matter appears to account for most of the difference in total organic C concentration among treatments (Table 17.7). Analysis of soil ^{137}Cs indicated that only minimum erosion had taken place and suggested that differences in organic matter among treatments were primarily attributable to variation in C inputs and decomposition rates (Bremer et al., 1995).

IV. SITE 3: INDIAN HEAD, SASKATCHEWAN

A. Background

Of the 30 million ha of cultivated land on the Canadian prairies, about 50% is located in the Black soil zone (Udic Borolls). There are two ongoing long-term (> 30 years) crop rotation studies being conducted in this zone, one at the Agriculture and Agri-Food Canada Experimental Farm at Indian Head, Sask. The Black soils occur in the fescue prairie–aspen grove (parkland) and the prairie grassland areas. These soils are very fertile, with surface horizons averaging 20–25 cm thick and organic matter content averaging 7%. Spring wheat is the main crop grown, but because favourable moisture regimes normally prevail, crop types are more diverse than in drier areas. For example, barley (*Hordeum vulgare* L.) and canola (*Brassica napus* L. and *Brassica camprestis* L.) are commonly grown. Mixed farming (both beef cattle and hogs) is quite common; thus the crop rotations usually include forages for feed. A significant proportion of the land is summerfallowed, primarily to control weeds and diseases. However, because of the contribution of fallow to soil degradation, producers have been encouraged to reduce summerfallow frequency, and there has been a trend in this direction in recent years (Campbell et al., 1986).

The crop rotation experiment at Indian Head was initiated in 1957 (1958 was first year of yields), primarily to study crop production and economics. Since 1987, Campbell et al. (1991a, d; 1993a, b; 1994) have conducted studies to evaluate the long-term effects of the various treatments on various soil attributes. We have chosen to document this study because the soil organic matter content is moderately high relative to other Black Chernozems, and the driving variables for crop production are near optimum; thus, the impact of agricultural management practices on soil quality should be strongly exhibited at this site.

B. Climate and geography

Indian Head is located in southeastern Saskatchewan, Canada, at 50°32′N and 103°31′W. The climate is subhumid continental, with a mean annual temperature 2.5 °C. The mean annual precipitation is 427 mm, and the mean annual Etp is 607 mm, giving a moisture deficit of 180 mm (Campbell et al., 1990). The frost-free period is 110 days.

The soil is classified as an Indian Head heavy clay, a thin Black Chernozem. It developed on clayey lacustrine deposits underlain by till and is situated on gently undulating topography showing slight evidence of wind erosion (Moss and Clayton, 1941). Virgin profiles of this soil had a granular dark brown to greyish black Ah horizon (0–15 cm) and a cloddy dark brown to black transitional A_2B_1 (15–30 cm) overlying a calcareous, massive, grey B_2 horizon. The organic C and N content of the 0- to 15-cm depth are 24.5 and 2.01 g kg^{-1}, respectively, the pH in water paste is 7.9, the average bulk density is 1.09 Mg m^{-3}, and field capacity is 42% by weight.

C. *Description of experiment*

Prior to commencement of this experiment, the site had been uniformly managed in a fallow–spring wheat (F–W) or F–W–W rotation, using conventional tillage management to control weeds (Zentner et al., 1986; Campbell et al., 1991a, d).

The experiment involves 11 crop rotations of which we discuss nine (Table 17.8). The plots were organized in a randomized block design with four replicates. All phases of each rotation were present each year. All tillage and cultural operations were performed with field-sized equipment. Sweet clover (*Melilotus officinalis* L.) as green manure (GM) and alfalfa (*Medicago sativa* L.)–bromegrass (*Bromus inermis* Leyss) as hay crops (H) were undersown with the preceding wheat crop.

Rates of N and P fertilizer applied to designated rotations varied over the study period (Zentner et al., 1986). During 1958 to 1977, fertilizer N and P were applied according to rotation specifications and the generally recommended rates for this region (Saskatchewan Soil Testing Laboratory, 1990), but thereafter N and P were

TABLE 17.8

Crop rotations[a] and fertilizer treatments[b] at Indian Head showing influence on grain production[c] per rotation per year (1958–1990)

Rotation	Average fertilizer application to rotation (1958–1990)		Grain production
	N	P	
	(kg ha^{-1} yr^{-1})	(kg ha^{-1} yr^{-1})	
F–W	0	0	1088
F–W	3.9	5.2	1240
F–W–W	0	0	1133
F–W–W	15.9	6.4	1540
F–W–W (straw removed)	15.9	6.4	1558
GM–W–W	0	0	1315
F–W–W–H–H–H	0	0	1515
Cont W	0	0	1029
Cont W	45.4	9.0	1960

[a]F = fallow; W = spring wheat; GM = sweetclover green manure; H = alfalfa-bromegrass cut for hay; Cont = continuous.
[b]N and P applied based on general recommendations for crop and soil zone from 1960–1977, then based on soil test from 1978.
[c]These values include the fallow year (e.g., for F–W the value is yield of wheat ÷ 2). The hay yields were excluded from the calculation for the six-year rotation.

applied based on soil tests. During the period 1958 to 1977, average annual rates of 6 kg N ha^{-1} and 12 kg P ha^{-1} were applied to wheat grown after fallow; for wheat after wheat stubble the rates were 24 kg N ha^{-1} and 9 kg P ha^{-1}. During the period 1978 to 1991, wheat following fallow received 10.5 kg N ha^{-1} and 8 kg P ha^{-1}, and wheat following wheat 76 kg N ha^{-1} and 9 kg P ha^{-1}.

In 1987, and at various times thereafter, soil samples were taken from the 0- to 7.5- and 7.5- to 15-cm depths and analyzed for the soil quality attributes discussed earlier for the Swift Current study (Campbell et al., 1991a–d; 1993a–c; 1994). In 1991, soil samples were taken in spring and fall to determine possible leaching of nitrate and phosphate, measured to a depth of 4.5 m (Campbell et al., 1993a; 1994). Soil aggregate stability, assessed by dry and wet sieving, was analyzed for soils taken in spring and fall 1991 (Campbell et al., 1993c).

D. Results and discussion

1. Yield trends and grain production

Yield trends (five-year running mean) for wheat grown on fallow, or partial fallow in the GM system, responded positively to fertilizers and legumes (Campbell et al., 1993b), but there were no consistent trends with time over the period of study (data not shown). It is likely that N and P mineralized during the fallow period would tend to obscure any nutrient shortfalls even if there was a gradual depletion in soil fertility as a result of frequent fallow or failure to apply fertilizer. Grain yields of wheat grown on stubble show an upward trend for fertilized systems and a downward trend for unfertilized systems, partly reflecting the effects of fertilization on soil quality changes (Campbell et al., 1993b). Even rotations that included legumes, although resulting in higher stubble wheat yields (wheat grown on stubble) compared to unfertilized monoculture wheat yields (Fig. 17.5), had yields that followed a downward trend (GM–W–W) or remained constant (rotations that included hay). Campbell et al. (1993b) suggested that this was partly because these two rotations reduce available P in the soil (Table 17.9) and because GM–W–W may reduce available water in subsequent stubble wheat crops in drought years.

The amount of grain produced by wheat per rotation per year was directly proportional to cropping frequency when fertility was not limiting; it was increased by green manure and even more so by including hay in the rotation (Table 17.8). Straw removal from the F–W–W (N, P) rotation did not affect grain production.

2. Soil biochemical characteristics

The impact of the rotations and cultural treatments on various soil biochemical attributes has been reported (Campbell et al., 1991a, c, d; Janzen et al., 1992). Some of the main findings are discussed below.

Quantity of soil organic matter: The amount of soil organic C and N in the surface soil was directly related to cropping frequency (Table 17.9). Cont W had largest

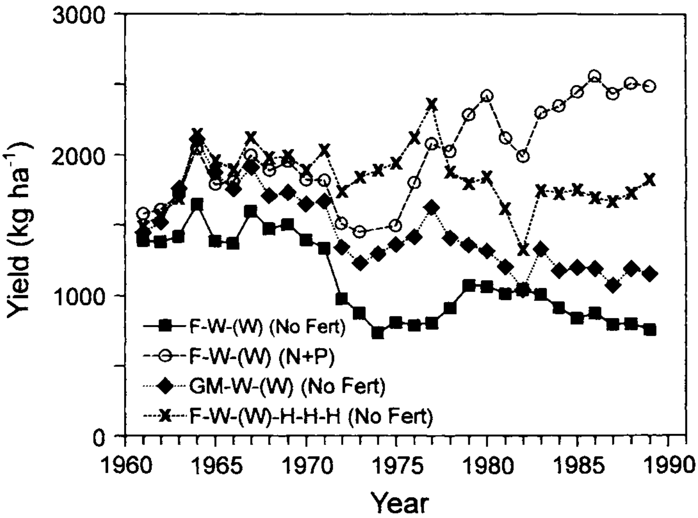

Fig. 17.5. Yield trends of hard red spring wheat grown on stubble in a fallow–wheat–wheat rotation on a thin Black Chernozem at Indian Head, Sask., showing the influence of sweetclover green manure, legume–grass hay crops, and fertilizer. All points are five-year running means (from Campbell et al., 1993c).

amount of organic C and N, whereas fallow–wheat systems had the smallest amounts. Fertilization increased organic C and N ($P < 0.05$) for Cont W but not for rotations that included fallow. Including sweet clover green manure or brome–alfalfa hay in cereal rotations increased soil organic C and N compared to the two- and three-year fallow–cereal rotations, but amounts of organic C and N were similar to those in fertilized Cont W. Straw removal from the F–W–W system had no effect on yield (Campbell et al., 1991c) nor on organic C and N (Table 17.9). These results suggest that root residues are more important than straw in their contribution to the maintenance of soil organic matter (ibid). Consequently, these authors suggest that the removal of two-thirds of the straw from fields in the Black soil zone during years of above-average crop production may not be serious in terms of maintaining soil organic C, because most of this C is destined to be lost to the atmosphere via respiration. However, in semi-arid areas, such as the Brown soil zone, keeping sufficient residue cover on the soil surface is important to protect it from erosion and to trap snow and conserve water. Furthermore, in Gray Luvisolic soils, which form crusts quite readily, it is also important to conserve crop residues to protect the soil.

Although the results at Indian Head and Swift Current clearly show that soil organic matter can be increased by frequent cropping (i.e., reducing fallow frequency) and improving the fertility of the soil, this is unlikely to occur in cases where the soil organic matter is very high (Fig. 17.6). Campbell et al. (1991b) stress that for the latter soils all we can hope to accomplish with good crop management is to reduce the rate of soil degradation.

TABLE 17.9

Selected soil biochemical and chemical characteristics of a thin Black Chernozemic soil at Indian Head, Sask., as influenced by crop rotation and fertilization (1958–1990). Values represent the 0- to 15-cm depth, except for P which is for 0- to 20-cm (from Campbell et al., 1991a, c, e, and 1993b; Janzen et al., 1992)

Rotation phase sampled[a]	Fertilizer[b]	Organic Matter[c]		Hydrolyzable Amino-N[d]	Microbial biomass		Light Fraction		Mineralization[e]		Olsen-P[f]
		C	N		C	N	C	N	C	N	
		(t ha^{-1})		(t ha^{-1})	—(kg ha^{-1})—				(kg ha^{-1} wk^{-1})		(kg ha^{-1})
(F)–W	N, P	30.1	2.87	1.04	815	75	627	26	178	32	ND
(F)–W		30.7	3.07	1.02	905	79	888	32	184	25	ND
(F)–W–W	N, P	31.9	3.40	1.12	891	88	687	28	193	38	55
(F)–W–W(-Straw)	N, P	30.4	3.04	1.05	833	80	643	26	168	32	ND
(F)–W–W		31.0	3.17	1.01	804	89	912	31	200	28	37
(GM)–W–W		36.5	3.56	1.15	987	125	811	37	214	44	26
(F)–W–W–H–H–H		36.0	3.72	1.27	1074	128	1390	70	227	49	24
Cont (W)	N, P	37.9	3.68	1.22	1029	124	1315	63	230	55	42
Cont (W)		32.1	3.28	1.10	938	96	800	35	190	36	31
Significance		**	**	**	**	**	**	**	**	**	**
LSD ($P < 0.10$)		3.7	0.29	0.10	115	22	374	17	22	8	14

[a]Treatment designations are: F = fallow, W = spring wheat, Cont = continuous, GM = green manure (sweetclover), H = hay (bromegrass & alfalfa), -straw = about two-thirds straw removed at harvest. The rotation phase that was sampled is given in parentheses.
[b]N and P applied primarily at rates based on soil tests made in fall.
[c]The previously published values for organic C were actually total C, measured using a dry combustion procedure. These values are measurements made on the same samples using an elemental analyzer (Carlo Erba, Milan, Italy)(Campbell, unpublished data).
[d]Amino acids plus amino sugars.
[e]Carbon mineralization determined by incubating field moist soil wetted to field capacity, at 20 °C for 14 d. Nitrogen values shown are the initial potential rate of N mineralization (N_0 X k) determined by incubating air-dry soil that was rewetted and incubated at 35 °C for 16 wk, with mineral N formed being extracted by intermittent leaching (Campbell et al., 1991c).
[f]$NaHCO_3$ - extractable P_i, on samples taken on 4 September, 1990. All other characteristics in this table based on samples taken in June, 1987.
ND = Not determined.

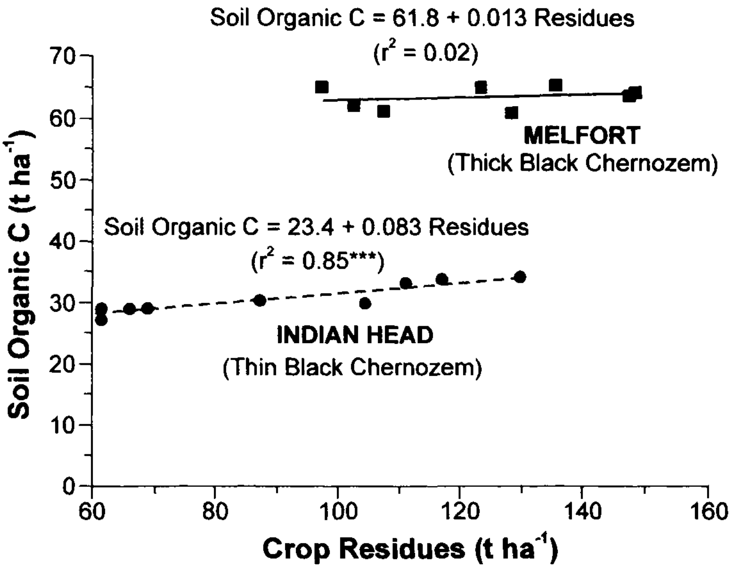

Fig. 17.6. Relationships between soil organic C in the 0- to 15-cm depth and estimated crop residues (including roots) returned to Melfort soil (31 years) and Indian Head (30 years) (adapted from Campbell et al., 1991b). Note: relationship for C revised based on correct values for organic C (see Table 17.9).

Amino compounds in soil: Campbell et al. (1991e) found that the hydrolyzable amino N was only marginally more effective than Kjeldahl N in identifying changes in soil organic matter quality (Table 17.9). Fertilization, frequent cropping, and including legumes as green manure or hay in rotations significantly ($P < 0.10$) increased amino N (amino acid plus amino sugar N). The effect of fertilization became greater as cropping frequency was increased. When the relative molar distribution of the amino acids was used to assess changes in soil organic N quality, increases in aspartic acid and decreases in arginine and leucine were observed. These differences were mainly associated with the high organic N systems (i.e., the six-year mixed and fertilized Cont W rotations). These results contrast with findings at Swift Current, where the treatments were assessed after half as long and lower response to management was observed.

Soil microbial biomass: Microbial biomass C (MB-C) and N increased (especially in the 7.5- to 15-cm depth) with increasing frequency of cropping (Table 17.9) and with the inclusion of legumes as green manure or hay in rotation (Campbell et al., 1991a). The influence of treatments on MB-C was less pronounced than on microbial biomass N (MB-N), which is similar to the findings of Powlson et al. (1987). For example, the MB-C on Cont W (N + P) was 26% greater than F–W (N + P), whereas MB-N was 65% greater. The influence of the treatments on microbial biomass C/N ratio indicated that the suite of microbial organisms may have been altered by treatments that increased soil organic matter (Campbell et al., 1991a). Microbial biomass C and N were positively correlated ($r = 0.70$, $P < 0.05$, for C and

0.81, $P < 0.01$, for N) with the estimated amount of crop residues (including roots) returned to the soil over the period of study (Campbell et al., 1991a).

Light fraction organic matter: The LF-C accounted for 2.0 to 5.4% of the soil organic C, and LF-N constituted 1.2 to 2.8% of the soil organic N (Janzen et al., 1992). Light fraction C and N contents were generally lowest in rotations with a high frequency of summerfallow, and fertilization of Cont W and inclusion of hay in the rotation favoured light fraction accumulation (Table 17.9). Light fraction was the attribute most sensitive to changes in SOM; for example, LF-C in Cont W (N, P) was more than two times higher than that in F–W (N + P). Straw removal in the F–W–W (N, P) system had no effect on LF-C or -N. Light fraction N content was correlated to microbial biomass N ($r = 0.61$, $P < 0.01$), suggesting that light fraction is a useful indicator of labile organic matter.

Mineralizable C and N and specific respiratory activity: Generally, C and N mineralization increased with cropping frequency, with fertilization of Cont W, and with the inclusion of legumes (especially hay crops) in the rotation (Table 17.9). Thus, laboratory measurements of C and N mineralization appear to be sensitive indicators of the effects of rotation and crop management practices on soil organic quality. The initial potential rate of N mineralization (N_0k; Table 17.9) was also shown to be sensitive to changes in soil quality. For example, in a growth chamber study in which soils from six of these cropping systems were used to determine the response of spring wheat to various rates of N and P fertilizer, the results showed a very close correlation between grain yield of wheat and N_0k ($r = 0.97^{***}$) (C.A. Campbell, unpublished data). Straw removal at harvest significantly ($P < 0.10$) reduced C and N mineralization (Table 17.9). There was a close relationship between N_0k and C mineralization ($r = 0.92$, $P < 0.01$), MB-N ($r = 0.68$, $P < 0.01$), and LF-N ($r = 0.60$, $P < 0.05$). There was also a significant correlation between C mineralization and LF-C and -N ($r = 0.79$, $P < 0.01$) and MB-C ($r = 0.50$, $P < 0.05$) (Janzen et al., 1992). These results are not surprising because light-fraction material and microbial biomass act as substrate for mineralization of C and N.

Multiple regression showed that MB-C and LF-C accounted for 82% of the variability in C mineralization ($n = 9$). The relationship for the top 15-cm depth (all values in kg ha^{-1}) was:

$$C_{min} = 69 + 0.10 \text{ MB-C} + 0.044 \text{ LF-C} \qquad (2)$$

The LF-C was significantly related to C mineralization at $P = 0.001$, as was MB-C at $P = 0.05$. Thus, as found at Swift Current, both light fraction and microbial biomass appear to contribute to C mineralization independently. The standardized estimate of the parameters in equation 2 were 0.40 for MB-C and 0.57 for LF-C, indicating that the influence of both these factors on C mineralization was almost equal in this soil. These results cannot be directly compared with those at Swift Current because of differences in the soil depth considered and in crops in the rotations.

Although we observed some significant treatment effects on specific respiratory activity in other soils (Campbell et al., 1992a,b), data for the top 15 cm of the Indian Head soil did not provide similar evidence (Campbell et al., 1991c).

All of the soil biochemical characteristics discussed here were closely correlated with the estimated crop residues (including roots) returned to the soil over the period of study (Campbell et al., 1991c). As observed at Swift Current and Lethbridge, there was a close and direct relationship between crop production (residue production; kg ha^{-1} yr^{-1}) and the maintenance of soil organic matter quality.

If we estimate the sensitivity of the various biochemical attributes to agronomic variables as the ratio of the values obtained for Cont W (N + P)/F–W (N + P), as was done for the Swift Current study, we find the ranking for the 0- to 15-cm depth to be: LF-N > LF-C > $N_{min} \approx$ MB-N $\geq$ $C_{min} \approx$ MB-C > amino-N ≈ soil N > soil C.

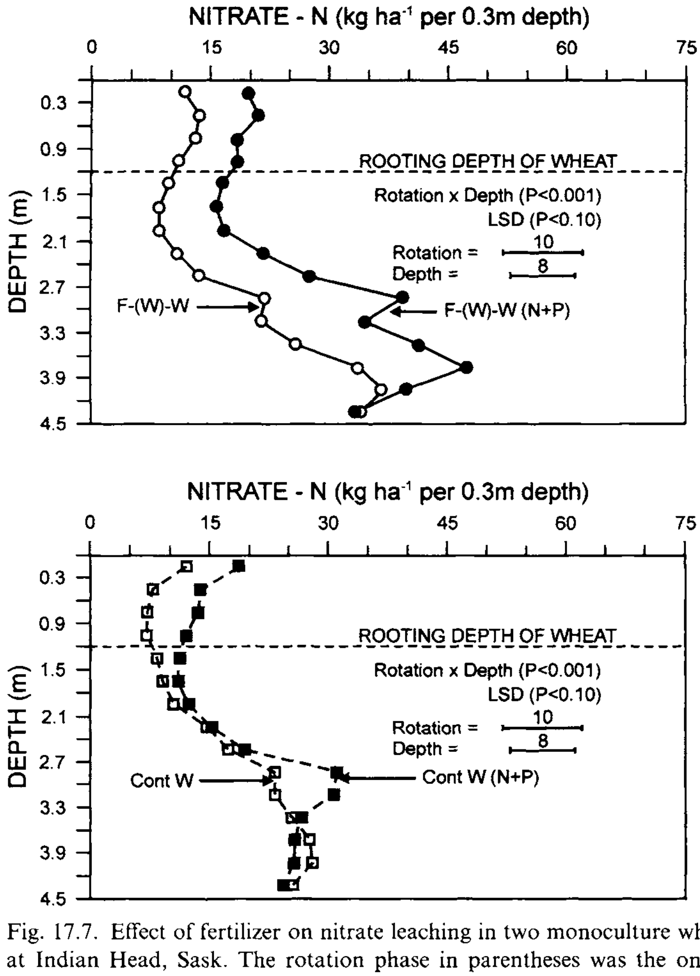

Fig. 17.7. Effect of fertilizer on nitrate leaching in two monoculture wheat rotations after 34 years at Indian Head, Sask. The rotation phase in parentheses was the one sampled (from Campbell et al., 1994).

Thus, as at Swift Current the most sensitive attributes appear to be the light fraction organic matter, followed by the mineralizable material, microbial biomass, and gross organic matter. This confirms that the soil biochemical attributes are more dynamic than the total organic matter.

3. Soil chemical and physical characteristics

The influence of the rotation treatments on soil available P (Olsen-P) and NO_3–N distribution in and beyond the root zone was assessed by taking measurements to 4.5-m depth, in May and September 1991 (Campbell et al., 1993b; 1994). Soil aggregation was also assessed by sampling the top 5 cm of soil in spring and fall 1991 (Campbell et al., 1993c).

Available P: During the growing season, most of the change in Olsen-P occurs in the 0- to 1.2-m layer of soil due to plant uptake (Campbell et al., 1993b). Uptake was mainly apparent in the fertilized and in the legume-containing systems, perhaps because these systems have the greatest plant production and, thus, P uptake. Comparison of Olsen-P distribution in the soil in May and September showed that phosphate may have leached below the root zone of cereals, especially when fallow-containing rotations were fertilized with N and P (Campbell et al., 1993b). Further, the results indicated that deep-rooted legumes (green manure and hay crops) increased phosphate in the subsoil, possibly through root decomposition in situ.

Available N: Considerable nitrate leaching occurred (Fig. 17.7, top) where summerfallow constituted a significant component of the crop rotation, because

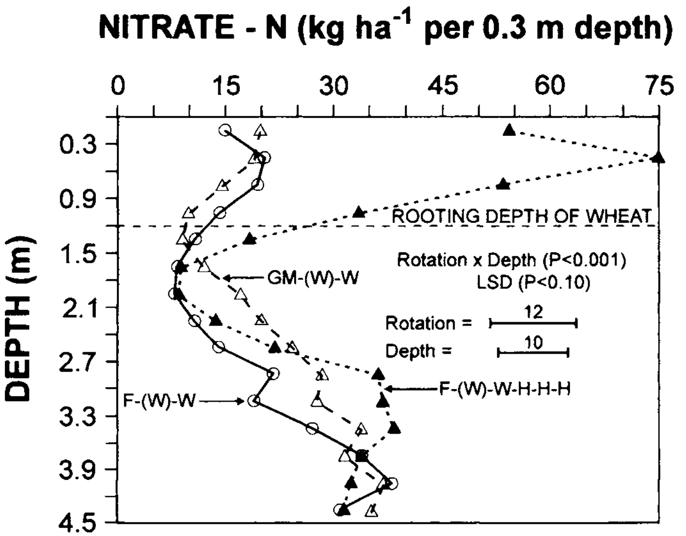

Fig. 17.8. Effect of legume green manure and legume–grass hay crops on nitrate leaching in rotations after 34 years at Indian Head, Sask. The rotation phase in parentheses was the one sampled (from Campbell et al., 1994).

this allowed storage of water in soil, thereby priming the soil for "leakage" (Campbell et al., 1994). Further, in the summerfallow period, N mineralization and nitrification predominates and, with plant N uptake being small, nitrate can be easily leached, especially during wet springs.

Both over-fertilization (Campbell et al., 1994) and under-fertilization (Campbell et al., 1993e) can lead to nitrate leaching. However, results in this rotation study showed that if fertilizer is applied based on soil test criteria and continuous cropping is practised, then nitrate leaching will be minimized (Fig. 17.7, bottom). There is a common misconception that, if soil available N is derived from legume decomposition, it will not contribute to nitrate leaching; however, results at Indian Head showed that this is not true (Fig. 17.8). The large amounts of nitrate leaching

TABLE 17.10

Effect of crop rotation and fertilization on GMD[a], GSD[a], and water-stable aggregates[b] at Indian Head, Sask., on two dates in 1991 after 34 years of crop rotation and fertilization (from Campbell et al., 1993d)

Crop Rotation	Fertilizer	3 June	24 Sept	3 June	24 Sept	Estimated[c] crop residues of the preceding crop (i.e., for 1990)
		GMD (GSD)		Water-stable aggregates (fast wetting)		
		— (mm) —		— (%) —		(kg ha^{-1})
(F)–W		2.3 (0.03)	19.0 (0.13)	47.0	38.6	5078
F–(W)		5.9 (0.06)	14.4 (0.08)	52.0	43.3	
(F)–W	N, P	23.8 (0.25)	30.6 (0.20)	52.9	44.0	8546
F–(W)	N, P	6.0 (0.06)	12.2 (0.12)	53.1	44.9	
(F)–W–W		9.9 (0.10)	10.8 (0.08)	53.9	47.0	2746
F–(W)–W		6.5 (0.04)	27.5 (0.13)	50.5	43.4	
(F)–W–W	N, P	20.9 (0.24)	23.1 (0.17)	56.3	51.0	7537
F–(W)–W	N, P	6.1 (0.02)	24.3 (0.20)	60.6	51.0	
(F)–W–W (Straw baled)	N, P	15.2 (0.20)	16.4 (0.13)	55.8	46.0	
(GM)–W–W		16.3 (0.17)	19.8 (0.16)	61.2	53.4	4741
(F)–W–W–H–H–H		13.7 (0.10)	18.0 (0.12)	74.1	71.8	
F–W–W–H–H–(H)		12.4 (0.25)	19.4 (0.22)	76.8	76.8	
Cont (W)		8.4 (0.01)	15.0 (0.14)	59.9	49.4	2624
Cont (W)	N, P	24.3 (0.16)	50.8 (0.24)	68.3	67.2	7759
[d]LSD ($P<0.05$) (Time × rotation phase):				Rotation phase × time = 4.8		

[a]Calculated according to Gardner (1956). No LSD calculated for GMD; GSD = Geometric standard deviation.
[b]Soil from fraction C of dry sieving (i.e., > 1mm < 2mm aggregates) used for wet-sieving analysis.
[c]Estimated from grain yields as outlined by Campbell et al., (1991a).
[d]LSD calculated for split plot with rotation-phase as main plot and time of sampling as sub-plot. The interaction of time X rotation phase was significant, therefore only LSD for assessing interaction means is presented.

observed under the legume systems were partly credited to the increase in N-supplying power of the soil that accompanied their use (Table 17.9), and to the summerfallow period following hay plowdown, or partial summerfallow following green manure plowdown, which facilitated soil water storage and easy leaching of nitrates in wet years (Campbell et al., 1994). We suspect that if there was no fallow or partial fallow, leaching in these systems would have been minimal.

These results emphasize the need to crop frequently, so as to make use of mineral N, and to apply fertilizer according to soil tests. They also show that excess available N has the potential to leach irrespective of its source, mineral or organic.

Soil aggregation: As at Swift Current, aggregate size distribution, determined by dry sieving, showed that summerfallowing increased the wind-erodible fraction of soil (< 0.84 mm) (Campbell et al., 1993c) and decreased the geometric mean diameter (GMD) of aggregates (Table 17.10). Fertilization, legume green manure and hay crops, and frequent cropping decreased the wind-erodible fraction of soil and increased the GMD and water-stable aggregates. The beneficial effect of the aforementioned treatments on soil aggregation was associated with their positive influence on crop residue production (Campbell et al., 1993c). Thus, factors that promote greater crop production are likely to contribute to improved soil aggregation, soil structure, and thus soil quality, in general.

V. SITE 4: BRETON, ALBERTA

A. Background

Most Gray Luvisolic soils in Canada occur in the northern interior plains of Manitoba, Saskatchewan, and Alberta and account for 5% of the 30 million ha of cultivated land on the Canadian Prairies. Gray Luvisolic soils occupy 20 million ha in Alberta. Of these, about 1.6 million ha are currently cultivated (Holmes et al., 1976), which is equivalent to 15% of the total cultivated area in Alberta (Bentley et al., 1971). Future expansion of arable agriculture in Alberta will be in areas dominated by Gray Luvisolic soils (Bentley et al., 1971).

Gray Luvisolic soils develop under forest vegetation. They have a thin A horizon that has a slightly acidic reaction, low organic C content, and low nutrient-supplying capacity. The A horizons often develop poor tilth when cultivated. Most soil-related problems, such as crusting, low water-holding capacity, low fertility, and low capacity to buffer against pH change, arise from the low organic matter content of the A horizon. Impeded water transmission and restricted root growth result from the presence of a dense, very firm, and acidic B horizon. These soils are currently used for grain, oilseed, and forage production.

The Breton Plots, managed by the Department of Soil Science at the University of Alberta, were established in 1930 on dominantly Gray Luvisolic soils (Typic Cyroboralfs) to find "*a system of farming suitable for the wooded soil belt*" (Robertson, 1979). Highlights of research conducted have been prepared by Robertson (1979, 1990). The Breton Plots are unique because all above-ground production has been removed since 1930. We chose to document this study because proper management of these soils leads to accumulation of soil organic matter and marked improvement of the soil physical, chemical, and biological properties and crop production. In addition, these changes are mainly due to below-ground crop residues and/or inorganic and organic amendments.

B. Climate and geography

The Breton plots are located 110 km southwest Edmonton (53°07′ N, 114°28′ W), 5 km from the town of Breton. On average, Breton receives 547 mm of precipitation each year, with 405 mm as rain (Atmospheric Environmental Service, 1982a, b). The potential evapotranspiration for this site is 420 mm; therefore available moisture is rarely a concern. The months of greatest rainfall are June, July, and August, and the greatest snowfall occurs during December and January. July is the warmest month, with an average minimum temperature of 8.8 °C and a maximum of 21.2 °C. January is the coldest month, with an average minimum temperature of −19.5 °C and an average maximum temperature of −8.6 °C. The average annual maximum temperature is 8.2 °C and the average annual minimum temperature is −4.1 °C. Breton has an average of 80 frost-free days and about 1150 growing degree days >5 °C.

The soil at the experimental site is classified as Breton Loam. The soils of the area developed on sandstones and shales of freshwater origin (Paskapoo formation). Soil formation was influenced by native vegetation consisting primarily of stands of white poplar (*Populus tremuloides* Michx) and poplar–spruce (*Picea glauca* (Moench) Voss) combination. The land was settled and cleared circa 1920 and farmed prior to the preliminary trials of the formal experiment, which were started in 1929. The topography of the site is level to gently sloping (slopes 0 to 4%) with a southwest aspect. Organic C and N contents of the 0- to 10-cm depth are 22 and 1.85 g ka^{-1}, respectively; the pH in water paste is 6.2, and the average bulk density is 1.11 Mg m^{-3}.

C. Description of experiment

Preliminary fertilizer trials were conducted in 1929 and the experiment formally established in 1930. The experiment was designed to compare two cropping systems and test several soil amendments including fertilizers, manure, and lime (Table 17.11). Originally, the experiment consisted of five blocks of land (Series A–E) that accommodated the two cropping systems, across which ran 11 strips with the various soil amendments. In 1938, an additional block of land was added (Series F) to expand the four-year rotation to a five-year rotation. Further, in 1941 the continuous wheat system (Series E) was split in two to create the present day two-year fallow–wheat rotation. Only three of the 11 amendments are compared in this study: check, manure, and NPKS (treatments 1, 2, and 3).

The cropping systems studied were the two-year rotation and a five-year rotation consisting of spring wheat–oat (*Avena sativa* L.)–barley (*Hordeum vulgare* L.)–

TABLE 17.11

Crop rotations and treatments at the Breton plots in Alberta

Rotation	Average NPKS application rates from 1930 to 1979 ($kg\ ha^{-1}\ yr^{-1}$)		
	Check	NPKS[c]	Manure[c]
F–W[a]	0–0–0–0	10–6–16–10	76–42–91–20
W–O–B–H–H[b]	0–0–0–0	10–6–16–10	76–42–91–20

[a]The present day two-year wheat–fallow rotations is on Series E. These plots were in continuous wheat from 1930 to 1940 and were split in 1941.
[b]The present day five-year wheat–oat–barley–hay–hay rotation is on Series A, B, C, D, and F. The four-year wheat–oat–barley–hay rotation established in 1930 was extended to the present five-year rotation in 1938. Forage species in the hay have varied over time: sweet/red clover (*Trifolium pratense* L.) of alfalfa/red clover from 1930 to 1955; alfalfa, red clover, creeping red fescue (*Festuca rubra* L.), bromegrass, and either Timothy (*Phleum pratense* L.) or alsike clover (*Tritolium hybridum* L.) from 1956 to 1966; and alfalfa and bromegrass from 1967 till the present.
[c]See text for details of these treatments.

forage–forage. The forage species have varied over time, consisting of legumes alone from 1930 to 1955 and legume/grass mixtures since 1956 (Table 17.11).

On average the NPKS treatments received 10, 6, 16, and 10 $kg\ ha^{-1}\ yr^{-1}$ of N, P, K, and S, respectively; manure treatment received 76, 42, 91, and 20 $kg\ ha^{-1}\ yr^{-1}$ of N, P, K, and S, respectively. The amount of N applied as inorganic fertilizer or manure was a function of the crop type and its place in the rotation (Juma, 1993).

Commercial fertilizer application methods varied over the years. Initially, fertilizers were broadcast annually, but during 1946–1963 fertilizers were broadcast every second year. In 1964, annual applications resumed and phosphate was drilled with the seed. In 1972, the east half of the five-year rotation plots (Series A–D and F) and complete plots of the two-year rotation (Series E) were limed if the pH was < 6.5. Manure was applied at a rate of 44 $t\ ha^{-1}$, once per five-year rotation up to 1979. Nutrient rates are annual equivalents and are estimated from manure analyses from 1976–1986 inclusive. For the five-year rotation, the manure was applied at the time the forages were plowed under, whereas for the two-year rotation it was applied to both fallow and wheat portions of the rotation in the fall, once every five years (Juma, 1993).

Amendment treatments were not randomized, and most were not replicated. Plot size was 31.5 m $\times$ 8.5 m, except in the two-year rotation for which plots were half this size.

D. Results and discussion

1. Grain and dry matter yields and yield trends

The crop yields in the two-year and five-year crop rotations have been reported (Juma, 1993; 1995). Soil sampling was last conducted in 1979; therefore crop yield data and soil N in different rotations and management practices up to 1979 are presented here.

Average grain yields in the NPKS- and manure-treated plots were generally similar and were up to 87% higher than yields of the check plots (Table 17.12). In the five-year rotation, average grain yields of wheat ≅ oat > barley. The benefit of the legumes in the forage did not appear to carry forward to the third cereal crop in the rotation, and barley yields were thus considerably lower than wheat yields. Wheat grain yield from NPKS and manure treatments of the five-year rotation were consistently higher than those of the two-year rotation. This suggests that the synergistic effect of legumes and manure or fertilizer on wheat yields in the five-year rotation was greater than in the two-year rotation.

In these experiments the first-year forages were harvested for hay in early to mid-July and the second-year forages were "plowed down" in late July without much regrowth. The trends in the total above-ground dry matter yields were similar to those of the grain yields (Table 17.12). The forage yields in the check and manure treatments were lower than those of above-ground yields of barley, but they were higher than barley yields in the NPKS treatment. This may be due to the addition of S in this S-deficient Luvisolic soil. The yields of forages in the NPKS- and manure-treated plots were almost two- to three-fold higher than those of the above-ground yields in the check plots. The amounts of N, P, K, and S added through manure were several times larger than those added in the fertilizers, but the S from the manure may not have been as available as that from the fertilizer.

It is important to note that fertilizer N in the NPKS treatment was applied at a rate of 10 kg ha^{-1} yr^{-1}. The total above-ground dry matter yield for cereal crops on these plots ranged from 3100 for barley to 6425 kg ha^{-1} for wheat (Table 17.12). Above-ground yield for fertilized barley at Breton, when currently accepted

TABLE 17.12

Long-term average grain and above-ground dry matter yields and their standard deviations at the Breton Plots, Alta. (1930–1979) (adapted from Juma, 1995)

Rotation	Crop	Treatments		
		Check	NPKS	Manure
		— (kg ha^{-1}) —		
Grain yields				
2-yr	Wheat	1225 (525)	1625 (775)	1925 (825)
5-yr	Wheat	1200 (525)	2250 (875)	2200 (800)
	Oat	1250 (525)	1925 (700)	2075 (800)
	Barley	825 (450)	1350 (525)	1500 (675)
Dry matter yields				
2-yr	Wheat	3375 (1425)	4500 (1950)	5500 (1850)
5-yr	Wheat	3050 (1175)	6425 (2350)	5750 (2025)
	Oat	3225 (950)	5150 (1700)	5575 (2100)
	Barley	1925 (800)	3100 (1100)	3400 (1150)
	Hay	1350 (825)	4000 (2350)	2950 (1250)
	Hay	1275 (1000)	4225 (1975)	3000 (1525)

technologies and N fertilizer rates (80 kg ha^{-1} yr^{-1}) are used, is 7250 kg ha^{-1}. This suggests that the long-term addition of fertilizer nutrients, combined with the use of crop rotations that include legumes, may be building the fertility of this soil. Analysis of data presented in Table 17.12 also showed that the straw C/grain C ratio of wheat and oat ranged from 1.54 to 1.85, and that for barley ranged from 1.26 to 1.33. Although the above-ground crop residues were removed, the straw C/grain C and the shoot C/root C ratios may have a bearing on the amount of photosynthates transferred below ground and consequently on the amount of soil organic matter in this soil.

2. Soil biochemical characteristics

The impact of the rotations and cultural treatments on various soil biochemical characteristics have been assessed at Breton (McGill et al., 1986; Juma, 1995). We highlight some of the main findings here.

Quantity of soil organic matter: Linear regression showed that the slope of total N vs. time (years) in the three nutrient treatments of the two-year rotation were not significantly different from zero for the period up to 1979, indicating that there were no measurable changes in total soil N over 42 years (Table 17.13). In contrast, soil N increased with time in all nutrient treatments of the five-year rotation. The increased organic matter was the highest for the manure treatment followed by NPKS and check treatments. After 42 years, the soil organic N content was about 37%, 42%, and 42% higher in the soil of check, NPKS, and manure treatments, respectively, of the five-year rotation plots than in those of comparable two-year (wheat–fallow) rotation plots (data not shown). In a recent up-to-date analysis of the trends in organic C in these plots (up to 1993), it is seen that the effects of manure are significant ($P < 0.06$) even in the two-year rotation (Table 17.14, Fig. 17.9). The results for the five-year rotation confirm those obtained previously for organic N. The main factor affecting organic C and N in these treatments was the input of C and N during the growing season through roots and root products. Crops were

TABLE 17.13

Trends of total soil N (g kg^{-1} soil) in the 0- to 15-cm depth of the check, manure, and NPKS treatments of the two-year and five-year rotations at Breton, Alta. (1938–1979) (adapted from Juma, 1993)

Rotation	Treatment	Equation	r^2
2-yr	Check	Total N = 1.04 − 0.0018 × Year	0.16 ns
	NPKS	Total N = 1.09 − 0.0007 × Year	0.02 ns
	Manure	Total N = 1.14 + 0.0069 × Year	0.47 ns
5-yr	Check	Total N = 0.94 + 0.0094 × Year	0.89 **
	NPKS	Total N = 0.88 + 0.0163 × Year	0.85 **
	Manure	Total N = 0.93 + 0.0248 × Year	0.92 **

TABLE 17.14

Relationship between soil organic C (SOC) concentration in the 0- to 15-cm depth and time (1938–1993) in Breton plots (from Izaurralde et al., unpublished data)

Treatment	Equation	r^2
F–W	SOC = 1.1	$r^2 = 0$
F–W (NPKS)	SOC = −5.0 + 0.003 × Year	r^2 = (ns)
F–W (manure)	SOC = −20.9 + 0.011 × Year	$r^2 = 0.64$ ($P < 0.06$)
W–O–B–H–H	SOC = −20.9 + 0.011 × Year	$r^2 = 0.86$ ($P < 0.01$)
W–O–B–H–H (NPKS)	SOC = −27.7 + 0.015 × Year	$r^2 = 0.90$ ($P < 0.01$)
W–O–B–H–H (manure)	SOC = −52.9 + 0.028 × Year	$r^2 = 0.96$ ($P < 0.01$)

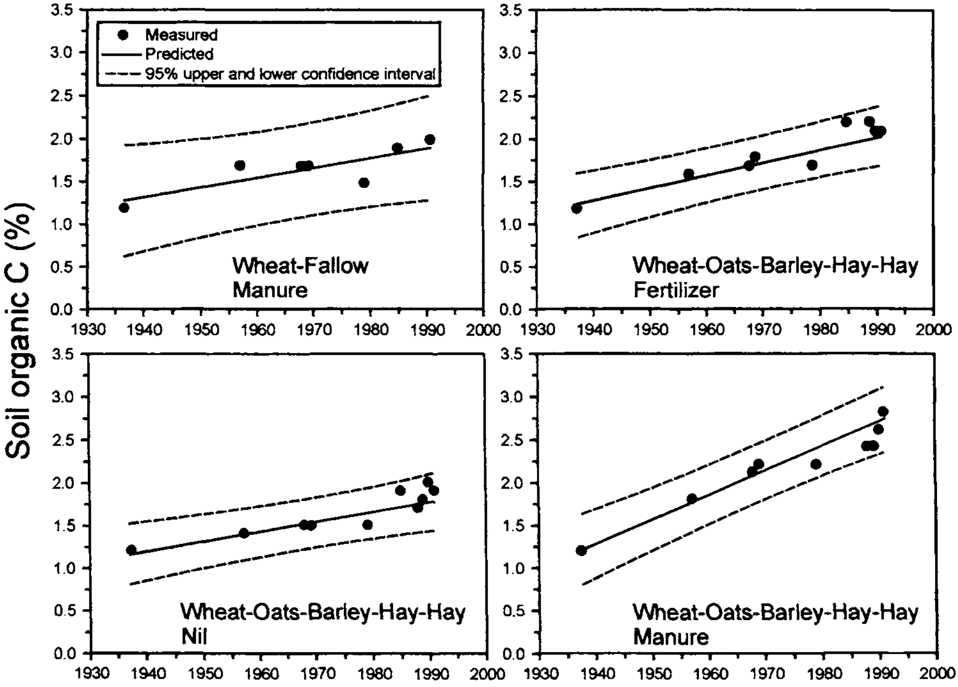

Fig. 17.9. Relationship between soil organic C and time (year) for the two-year and five-year rotations amended with NPKS or manure or unamended at Breton plots, Alta. (1938–1993) (from Izaurralde et al., unpublished data).

grown continuously in the five-year rotation, and therefore more plant material (stubble and roots) was added than in the two-year fallow–wheat rotation. The five-year rotation plots were bare for a smaller portion of the time, so that decomposition processes were probably slower (soil drier for longer) and erosion losses were likely smaller. A greater amount of N was probably left by the legume-containing forage crop roots than cereal crop roots, and this would be conducive to higher soil organic matter contents.

After 42 years of cropping at the Breton Plots, the soil organic N content in the NPKS treatments of the two-year and five-year rotations was 6% and 10% higher than in their respective check plots (data not shown). These higher values can be explained by higher crop yields (forages and grain) on the fertilized plots. Thus, the amount of stubble and root material added to the soil was greater on fertilized plots, even though all above-ground material was removed. The soil organic N content in the manure treatments of the two-year and five-year rotations was 36% and 41% higher than in the respective check plots (data not shown). This result was partly due to the manure being organic matter and the average annual addition was approximately 9 t ha^{-1}. In addition, manure improved crop growth and hence organic matter additions via roots and stubble.

Amino compounds in soil: The proportion of the N in the surface soil (0- to 15-cm) that was hydrolysed by 6*M* HCl was greater for the five-year than for the two-year rotation (79.0 vs. 75.1%) (Khan, 1971). The same was true for amino acid-N (30.9 vs. 28.8%), amino sugar-N (10.4 vs 9.3%), and the unidentified-N (22.6 vs. 19.2%). In contrast, the acid hydrolysed ammonium-N showed the opposite effect (15.1 vs. 17.8%), as did the non-exchangeable ammonium-N (9.6 vs. 12.1%). Prolonged use of inorganic fertilizers did not change the relative amounts of amino acid-N, hydroxyamino acid-N, ammonium-N, and unidentified-N in the acid hydrolysate of soils, nor soil N present as amino sugar-N. Although, the relative amount of total hydrolysable-N in soils was not significantly different among fertilizer treatments, there was a significantly smaller percentage of total hydrolysable soil N in the NPKS and manure treatments than in the check plots.

Examination of the relative molar distribution of amino acids indicated that fertilizers had no effect on the distribution of amino acids in these rotations (Khan, 1971). As found at Indian Head, the amount of amino acids was higher in the plots under the five-year rotation than in the two-year rotation and was related to the organic matter content of the soils under the different crop rotations.

Analysis of soil organic matter and carbohydrate fractions showed that organic C, organic N, hexose, pentose, uronic acid, hexosamine, and hexosamine-N contents were significantly higher in soil samples from the five-year rotation than in the two-year rotation (Khan, 1969). The proportion of organic C in the form of hexose did not differ significantly between these two rotations, but the amounts of pentose, uronic acid, and hexosamine were greater in the five-year rotation. Application of manure and fertilizer (NPKS) increased the contents of organic C, organic N, and carbohydrate materials, as well as the proportion of organic C present as hexose and hexosamine. These results are consistent with those obtained for amino acids.

Organic matter fractions: When measured in 1968, the humic acid content in soils under the five-year rotation was significantly greater than those of the two-year rotation (Khan, 1969; 1970). The values for humic acid-C/fulvic-C, E4/E6 ratio, neutralization capacity, and carboxyl contents of humic material isolated from soils under the two-year rotation were greater than in the five-year rotation. These results suggest that the soil organic matter under the two-year rotation was more humified than that in the five-year rotation (Khan, 1969;70), and therefore would be expected to be less fertile as well.

Soil microbial biomass: Soil microbes and fauna, particularly protozoa and nematodes, control the mineralization of soil C and N (Clarholm 1984; Elliott et al., 1984; Rutherford and Juma, 1992). McGill et al. (1986) studied the dynamics of soil microbial C and N in the two long-term rotations and found that the amount of microbial biomass C in 1981 was significantly higher ($P < 0.05$) in the manured plots (571 mg kg^{-1}) than in the check (389 mg kg^{-1}). However, they found no difference between samples of the manured and NPKS treatments (431 mg kg^{-1}) or between the NPKS and check treatments. The five-year rotation contained 38% more total soil N but 117% more microbial biomass N than did the two-year rotation, and the manured treatments contained twice as much microbial biomass N (40.7 mg kg^{-1} reported as flush of N) as did NPKS (22.7 mg kg^{-1}) or check plots (20.7 mg kg^{-1}). The average turnover rate of microbial biomass under the wheat crop in the two-year rotation was 1.08 yr^{-1} for the manure treatment, 1.02 yr^{-1} for the NPKS, and 1.29 yr^{-1} for the check; in the five-year rotation the rate was 0.55 yr^{-1} for the manure treatment, 0.93 for the NPKS, and 1.33 for the check (McGill et al., 1986). These data suggest that the turnover of microbial biomass in the manure and NPKS treatments of the two-year rotation is faster than in the five-year rotation.

Fyles et al. (1988), studying the dynamics of microbial biomass and faunal populations in the oat and alfalfa plots of the five-year rotation, found that the average microbial biomass C in the 0- to 15-cm depth in the oat plot (1310 kg ha^{-1}) was greater than in the alfalfa plot (910 kg ha^{-1}). The average nematode population over the four-month period in 1984 was significantly greater in the oat plot (4.1×10^{10} ha^{-1}) than in the alfalfa plot (2.8×10^{10} ha^{-1}). Average microarthropod numbers were significantly greater in the oat plot (8.1×10^{8} ha^{-1}) than in the alfalfa plot (4.8×10^{8} ha^{-1}). These differences in soil organisms were attributed to differences in soil moisture regimes, input of carbon through roots (oat root mass, 2630 kg ha^{-1}, was significantly less than alfalfa root mass, 4830 kg ha^{-1}), and the size and extent of the rhizosphere. The differences likely reflect the short-term influence of different crop types and management practices in the rotation.

Water-soluble organic C: The water-soluble organic C represents C that is immediately available to microorganisms. McGill et al. (1986) measured the dynamics of water-soluble C in the 0- to 5-cm depth of the check, NPKS, and manure treatments of the two rotations throughout the 1981 and 1982 growing seasons. In both years, water-soluble organic C was significantly greater ($P < 0.05$) in the manure treatment than in the check or NPKS treatments. For example, in 1981 the amount of water soluble organic C for manure, NPKS, and check treatments was 36, 24, and 25 mg kg^{-1} soil (oven-dry basis), respectively; in 1982 the corresponding values were 31, 21, and 21 mg kg^{-1}. These trends were similar to those found for microbial biomass. These results show that changes in soil organic matter due to management are associated with changes in microbial biomass and labile soil organic matter pools; thus the latter attributes may be used as indicators of soil quality.

3. Soil chemical characteristics

Soil pH: After 27 years of cropping, soil pH was determined (water paste) in June 1957 (Toogood et al., 1962). The soil pH values were generally lower in the five-year rotation than in the two-year rotation. In the five-year rotation, the trend in soil pH was NPKS (5.5) < manure (5.9) < check (6.1). Similarly, in the two-year rotation, values were NPKS (6.0) < manure (6.3) < check (6.4). Similar results were later reported for these treatments by McCoy and Webster (1977) and by Cannon et al. (1984). Thus, there appears to be a gradual acidification of NPKS plots over time, but the effect of manure is inconclusive.

Available N: Soil tests for available N (0- to 15-cm depth) were conducted in June 1957 after 27 years of cropping. The amount of nitrate-N in the 50-year rotation followed the trend: manure = NPKS > check (Toogood et al., 1962). In the two-year rotation the trend was manure > NPKS = check. A similar trend in available N was also found when the plots were sampled in the 1980s.

Available P: Soil tests for available P (0- to 15-cm depth) were conducted in June 1957. In the five-year rotation the amount of Olson-P was greatest in the manure treatment (manure > NPKS = check), whereas in the two-year rotation it was NPKS > manure > check (Toogood et al., 1962). McKenzie et al. (1992) used the Hedley fractionation technique (Hedley and Stewart, 1982) to assess changes in the forms of soil P in the check and NPKS treatments of the two rotations. The more biologically available Pi fraction showed greater drawdown in the check plot of the more frequently cropped five-year rotation than in the two-year rotation. Continuous cropping in the five-year rotation slowed the breakdown and decline of the Bicarb-Po and NaOH-Po fractions and was associated with a slight increase of residual-P. Cropping with fertilization resulted in a 30–40% decline in total soil P. The addition of N and P fertilizer contributed to the Pi pools in both rotations. Nitrogen fertilizer, which increased crop growth and therefore increased soil organic matter, indirectly contributed to increased labile and moderately labile Po pools in both rotations, except for the NaOH-Po in the five-year rotation.

Available S: Total S, measured on soil samples taken from the 0- to 15-cm depth in fall 1966 and spring 1967, ranged from 94 to 190 mg kg^{-1}; in all cases it was greater in the five-year than in the two-year rotation (Robertson, unpublished data). The trends in total S content parallelled those of soil organic C and N. In the two-year rotation, the amount of total S in the manure-treated plot was greater than in the check and NPKS plots. In the five-year rotation, the amount of total S in the manure and NPKS treatments was similar and significantly greater ($P < 0.05$) than in the check plot.

The HI-reducible S and C-bonded S represent the main sources of plant available S. The HI-reducible S varied from 40 to 65 mg kg^{-1} soil and accounted for 35–45% of total S. There was no significant difference in HI-reducible S contents in the check plots of the two rotations, but HI-reducible S in the manure and NPKS treatments was greater in the five-year than in the two-year rotations (Robertson, unpublished data). The C-bonded S ranged from 12 to 34 mg kg^{-1} soil and accounted for 11 to

18% of total S (Robertson, unpublished data). C-bonded S in the check and NPKS treatments of the five-year rotation was greater than in the two-year rotation. There were no differences in C-bonded S in the manure treatments of the two rotations.

4. Soil physical characteristics

Soil aggregation: Mean weight diameter of aggregates was determined using a wet-sieving technique. The aggregates in plots under the five-year rotation were 1.6-fold larger than the size of those in the two-year fallow–wheat rotation; however, in contrast to results at Swift Current and Indian Head, there was no effect of fertilizer treatments on soil aggregation (Cannon et al., 1984). Aggregate size was greater in the five-year than in the two-year rotation. These results are associated with soil organic matter content, input of C through forages, and the absence of fallow. Tillage and reduced crop residue production during the fallow period likely contributed to the smaller size of aggregates in the two-year rotation.

VI. CONCLUSIONS

Biochemical, chemical, and physical attributes of soil quality were measured at four long-term experimental sites located in major ecological zones of the Canadian prairies (i.e., Brown, Dark Brown, thin Black Chernozemic and Gray Luvisolic soil zones), to determine how management (cropping frequency, summerfallowing, application of fertilizers, manures, and growing legumes) influences soil quality.

Some soil quality attributes effectively reflected the impact of management on crop productivity and soil fertility at the three sites with Chernozemic soils (Swift Current, Sask., Brown; Lethbridge, Alta., Dark Brown; Indian Head, Sask., thin Black). These attributes, including total organic matter, light fraction organic matter (partly decomposed residues), microbial biomass, C and N mineralization, specific respiratory activity (i.e., ratio biomass-C/C mineralization), and soil aggregation, were suitable candidates for a minimum data set. Changes in soil organic matter quantity and quality were related to residue inputs (and in turn to crop production), as well as to conditions governing residue decomposition (e.g., soil moisture). Because the Indian Head soil has only moderate amounts of organic matter and is located in a sub-humid climate where production response to management is usually good to excellent, this site most effectively demonstrated the usefulness of the various soil quality attributes to establish a minimum data set. We demonstrated the positive effects of fertilization, legumes, and frequent cropping, and the negative effects of summerfallowing, on soil fertility and soil aggregation, nitrate leaching, and crop production.

Some attributes (e.g., light fraction organic matter, C and N mineralization, specific respiratory activity, and wet aggregate stability) were especially effective in differentiating between management practices. However, use of a minimum data set is recommended to achieve the most reliable assessments of soil quality.

Several of the attributes evaluated on the Chernozemic soils were not tested on the Gray Luvisol at Breton, Alta. On the other hand, at Breton it was shown that soil fauna is another attribute that can be used effectively to delineate the effects of management practices on soil quality.

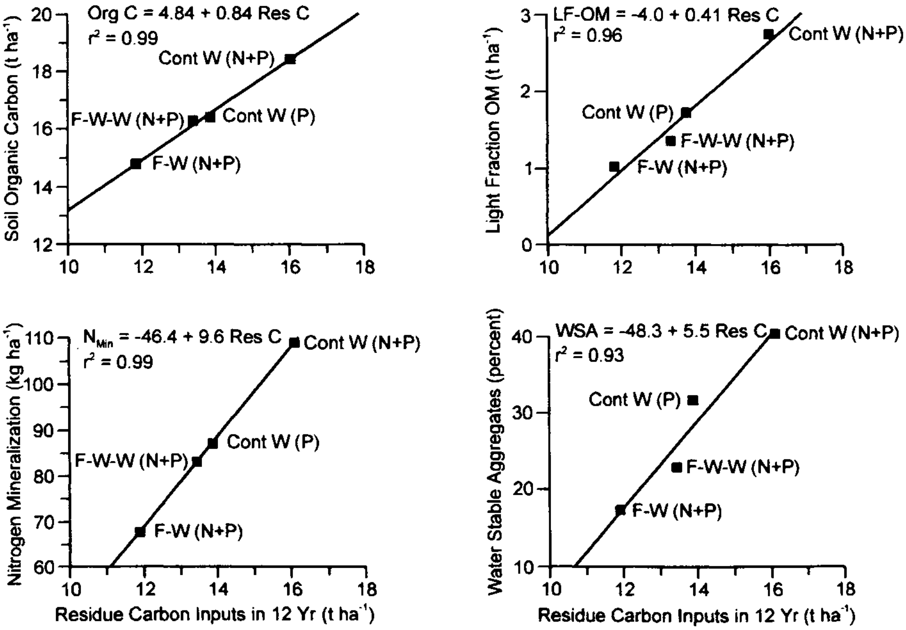

Fig. 17.10 The association between various soil quality attributes and crop residues based on results of the Swift Current long-term rotation study (adapted from Schoenau and Campbell, 1996).

Crop residues positively affected crop production through beneficial effects on soil, water, microorganisms and fauna and, in turn, enhanced crop production, thereby increasing residue production. This intimate feedback interaction implies that most steps taken to improve crop production are also likely to enhance soil quality. This desirable relationship is readily demonstrated in Figure 17.10, showing the interaction between crop residues and various soil quality attributes in the Brown Chernozem at Swift Current.

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Chapter 18

SOIL ORGANIC MATTER AND SOIL QUALITY—LESSONS LEARNED FROM LONG-TERM EXPERIMENTS AT ASKOV AND ROTHAMSTED

B.T. CHRISTENSEN and A.E. JOHNSTON

I. INTRODUCTION

The various elements of an arable cropping system, such as crop type and cultivar, soil tillage, crop protection measures, use of mineral fertilizers and animal manure, crop-residue disposal, and addition of town-derived organic refuses all interact in a complex manner with basic soil properties (texture and mineralogy), and this interaction determines the general productivity of the cropping system. The same elements also influence soil organic matter (SOM) turnover, some more than others. The SOM holds large pools of organically bound plant nutrients, which, through mineralization, may become available for crop uptake or loss from the soil–plant system through leaching or gaseous losses. Additionally, SOM is important to both soil structure and the potential for soil erosion, to sorption of mobile plant nutrients and retention of pesticides, and to the CO_2 balance between agroecosystems and the atmosphere. Thus SOM levels and turnover rates are intimately linked to soil properties of importance in maintaining an economically and environmentally sustainable agricultural production, and thereby also to soil quality.

Based on results from long-term field experiments conducted at Askov and Rothamsted experimental stations, we present in this chapter some of the lessons learned in relation to the role of SOM. We focus on the effects of cropping, applications of mineral fertilizers and animal manure, and crop-residue disposal on

SOM levels, and on the effect of SOM on crop yields and certain soil properties. The presentation is by no means exhaustive but exemplifies the research potential of long-term experiments.

II. SITE DESCRIPTIONS

The Askov Long-Term Experiments on Animal Manure and Mineral Fertilizers were initiated in 1894 on two sites of different soil texture. Sandmarken is a coarse sand (CS-soil) with 4% clay and 4% silt, whereas Lermarken is a light sandy loam (SL-soil) with 12% clay and 13% silt. A four-course rotation is grown on both sites, comprising spring-sown cereals, legume/grass mixtures, autumn-sown cereals, and root crops. Four fields (or blocks) are employed on each site (field B_2 to B_5 on Lermarken and field G_1 to G_4 on Sandmarken), allowing each crop to be grown every year. Lermarken had been cultivated for nearly a century when the experiments were initiated, and the cropping history of Sandmarken dated back at least two hundred years. Soil $pH_{(water)}$ is maintained at 6.0–6.5 through liming every four to five years. Annual precipitation is 860 mm and mean temperature 7.7 °C (1961–1990 averages). For a detailed description see Christensen et al. (1994).

We also draw upon results from three other experiments initiated at Askov in 1956. One of them deals with the effect of crop rotation on the SOM level, and was carried out next to the long-term experiments on Lermarken. The other two experiments were small-plot experiments confined in concrete cylinders (diameter 1 m, depth 0.5 m). One included different cropping systems, the other tested the effect of different organic amendments.

Rothamsted Experimental Station is on the outskirts of Harpenden, a small town some 40 km north of London. In 1843, J.B. Lawes and J.H. Gilbert started the first of a number of large-scale field experiments on the nutrient requirements of agricultural crops. Eight of the experiments started between 1843 and 1856 still continue, and these are collectively known as the Rothamsted Classical Experiments. Results from three of these are discussed here. Broadbalk, started in 1843, grows winter wheat each year, and Hoosfield Continuous Barley, started in 1852, grows spring barley each year. Barnfield, started in 1843, grew root crops each year until 1959 and now has different crops, but with the same manuring treatments. Results from the Rothamsted Ley-arable experiment are also discussed. Experiments were started in 1876 at Woburn Experimental Farm some 40 km north of Rothamsted. Thus the history of the sites of the two experiments discussed here are known in detail. These are the Woburn Ley-arable experiment, started in 1937, and the Woburn Market Garden experiment, started in 1942.

The soil at Rothamsted is a silty clay loam with 20–25% clay and 25–30% silt; soil $pH_{(water)}$ is maintained above 6.5 and annual rainfall averages 700 mm. The soil at Woburn is a sandy loam with about 12% clay and 24% silt; soil pH is maintained at 6.5 and average rainfall averages 600 mm. For a recent history of the Rothamsted Classical Experiments see Johnston (1994). Examples of other data from the experiments are given by Powlson and Johnston (1994) and Johnston and Powlson (1994).

III. AGRONOMY AND THE ORGANIC MATTER CONTENT IN SOIL

The agronomy practised on a soil is crucial to its SOM content. First we illustrate how growing crops with mineral fertilizers and animal manures and the use of other organic soil amendments like crop residues, sewage sludge, and peat affect SOM levels. Then we address how various crops and crop rotations influence SOM contents through differences in the quantity and quality of crop-residue inputs to soil, their seasonal distribution, and their ratios between above- and below-ground inputs.

A. Addition of mineral fertilizers, organic manures, and crop residues

1. Long-term trends

The main treatments of the Askov long-term experiments include different levels (½, 1, 1½) of animal manure (AM) and mineral fertilizers (NPK), and unmanured (0) plots. The level of nutrients added was altered in 1907, 1923, 1949, and 1973, but within each period almost the same amounts of total N, P, and K were applied in the corresponding AM and NPK treatments (Table 18.1). Until 1972, farmyard manure (FYM) was added to AM plots at a rate of 10 t ha^{-1} (wet weight) for 1 AM (annual mean for the crop rotation). During 1923 to 1972, AM plots also received liquid manure (LM, 4 t ha^{-1} to 1 AM). After 1972 FYM and LM was replaced by cattle manure slurry (25 t ha^{-1} to 1 AM) with 60% of the total N in ammoniacal form.

Since 1923 the soil has been sampled regularly at four to five year intervals; before then samples were taken occasionally. The lack of soil samples from the early years is a limitation that the Askov experiments share with several other long-term field experiments. However, it does not invalidate comparisons between treatments in the years when samples are available.

TABLE 18.1

Amounts of N, P, and K (annual mean of crop rotation) applied in NPK (mineral fertilizer) and AM (animal manure) treatments on both soils at Askov. The amounts were changed in 1907, 1923, 1949, and 1973

Period	Total nutrients added, kg ha^{-1} $year^{-1}$											
	1 AM			1½ AM			1 NPK			1½ NPK		
	N	P	K	N	P	K	N	P	K	N	P	K
1894–1906[a]	41	13	29	62	20	44	39	12	28	59	18	42
1907–1922[a]	42	13	33	63	20	50	42	13	32	63	20	48
1923–1948[b]	71	16	65	107	24	98	70	17	70	105	26	105
1949–1972[b]	93[c]	19	58	140[c]	29	87	70	18	66	105	27	99
1973–1994[d]	98	19	87	147	29	131	100	19	87	150	29	131

[a]AM was FYM.
[b]AM was FYM supplemented with liquid manure to root crops.
[c]AM supplemented with calcium nitrate (23 and 35 kg N ha^{-1} for 1 and 1½ AM, respectively).
[d]Since 1973 AM is cattle manure slurry having 60% of the total N in ammoniacal form.

Figure 18.1 shows the content of C and N in the plough layer (0–20 cm) in the B_3-field on Lermarken. SOM has declined in all soils, and the decreases are somewhat similar in all treatments. This could be ascribed to a continuing decline in the pool of "original" or native SOM, derived from the vegetation that preceded cultivation. This suggests that the effect of cultivating native soils in temperate regions may last for centuries, and the consequences for SOM contents should continue to be a matter of concern in relation to future soil quality. Differences

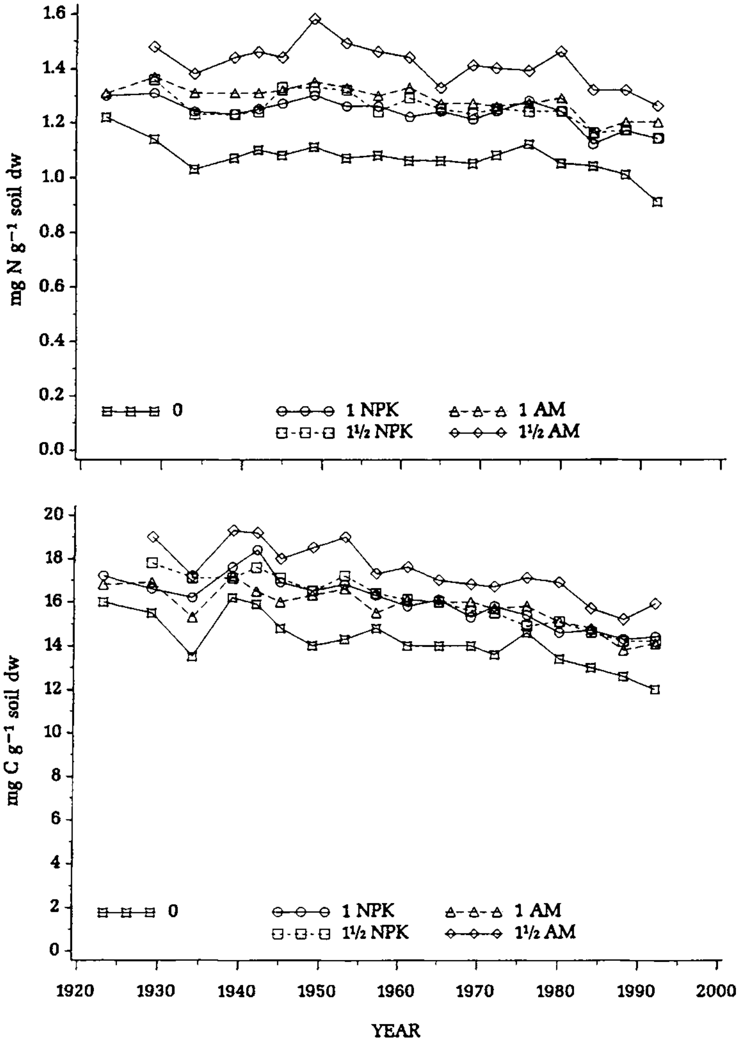

Fig. 18.1. The N and C content in the B_3-field on Lermarken, Askov. The treatments are: 0 = unmanured, NPK = mineral fertilizer, and AM = animal manure. NPK and AM were both given at level 1 and 1½ (see Table 18.1).

between treatments had been established by 1923, and those differences have changed little since then, suggesting that the equilibrium between addition and decomposition of SOM in each treatment was reached within 30 years.

Almost similar contents of SOM have been attained in soils treated with 1 AM, 1 NPK, and 1½ NPK. In contrast, unmanured and 1½ AM plots have lower and higher SOM contents, respectively. The higher SOM level in fertilized plots than in unmanured ones is ascribed to a greater crop productivity and hence a greater return of crop residues (rhizodeposits and stubble). Crop yields have been generally 2–3.5 times higher on 1 NPK- and 1 AM-treated plots than on unmanured ones, the greater yields usually being obtained with NPK. Similar amounts of plant nutrients have been added in 1 NPK and 1 AM (see Table 18.1), but the moderate extra organic matter input provided with animal manure has not raised the SOM level compared to 1 NPK. Similarly, SOM contents have not responded to increased rates of mineral fertilizers despite the significantly larger yields of the harvested crops obtained on 1½ NPK plots compared to 1 NPK (Christensen et al., 1994). This suggests that the return of rhizodeposits and other crop residues was similar. In contrast, plots getting 1½ AM have higher SOM levels. Taken together, the results suggest that animal manure does have the potential to raise SOM levels on this light-textured soil, but only when large amounts are applied. It should be recalled that animal slurry has been used since 1973 and that well managed, straw-rich FYM may be more beneficial to SOM build-up than slurry that is low in bedding materials and stored anaerobically.

Several factors have the potential to generate an overall decline in the level of SOM in arable soils. These include repeated soil tillage; the replacement of permanent vegetation with annual crops; and vertical transport of soil between the plough layer and subsoil, mediated by deep-burrowing earthworm species and by increased ploughing depth. If the depth of ploughing is fixed, an increase in soil bulk density will cause subsoil with less SOM to be introduced into the plough layer, resulting in a decrease in SOM. A higher bulk density may result from increased soil compaction following the introduction of heavier field implements. Horizontal transport of topsoil mediated by tillage operations, or wind or water erosion may also introduce SOM-depleted subsoil into the plough layer.

Figure 18.2 shows that on Hoosfield at Rothamsted, the SOM content has been constant for about 100 years on both the unmanured plot and that given NPK fertilizers. The quantity is a little larger in the fertilized soil, because larger crops have been grown and, although straw is removed each year, there have been larger residues from stubble, leaves, and roots returned to the soil. Annual additions of 35 t ha^{-1} fresh FYM have increased SOM, rapidly at first and then more slowly as equilibrium for this system is approached. But it is important to note that the time-scale over which this change has occurred is more than 130 years for this medium-textured soil in a temperate climate.

Figure 18.2 illustrates the fact that clay plays an important role in the retention of SOM and to equilibrium levels of SOM. The sandy loam at Woburn has about 12% clay, a clay content similar to that of the soil at Askov Lermarken, whereas the silty clay loam at Rothamsted has more than 20% clay. At the start of the long-term

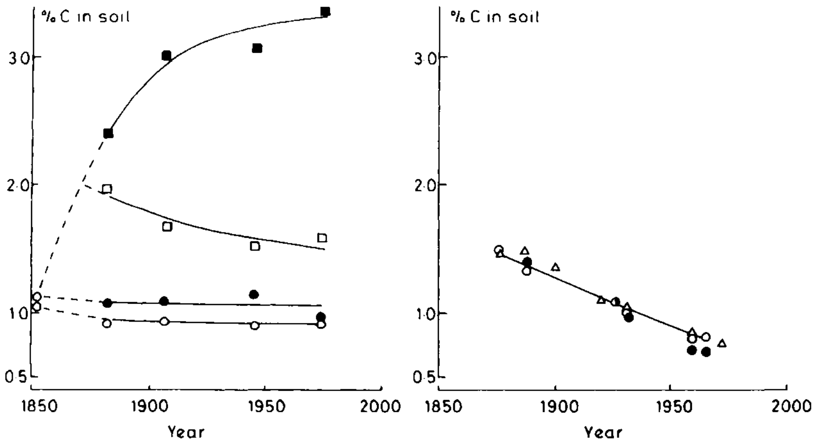

Fig. 18.2. Change in soil C content with time under all-arable cropping systems on a silty clay loam at Rothamsted (left), and a sandy loam at Woburn (right). Rothamsted: Barley grown each year, annual treatments since 1852: ○ unmanured; ● NPK fertilizers, 48 kg N ha^{-1}; ■ FYM 35 t ha^{-1}; □ FYM 1852–71, none since. Woburn: Cereals grown each year: ○ unmanured; ● NPK fertilizers; Δ manured four-course rotation (Jenkinson and Johnston, 1977).

experiments, SOM levels were higher at Woburn than at Rothamsted, but under continuous arable cropping there is now less SOM in Woburn soil than in Rothamsted soil. Johnston (1991) gave further examples of the effects of soil texture for other farming systems at the two sites.

The slow change in SOM content of a soil in a temperate climate is also well demonstrated by the changes in old arable soils containing about 0.1% N, which were sown to permanent grass at Rothamsted (Fig. 18.3). It took 100 years to achieve the apparent equilibrium level of SOM (corresponding to about 0.28% N), which is characteristic of permanent grassland in these conditions. The time required to reach the half-way stage between the two equilibrium values was 25 years (Johnston, 1991).

2. Medium-term aspects

The effect of the rate of addition of organic manures on SOM levels over shorter time-scales is illustrated in detail by results from the Market Garden experiment started on the sandy loam at Woburn in 1942. The experiment compared four organic amendments, each applied at either 37.5 or 75 t fresh weight ha^{-1}. Two of these were FYM and sewage sludge, the latter comprising a mixture of activated digested sludge and sludge taken directly from the drying beds following anaerobic digestion. The other two were FYM compost (FYM plus green vegetable material) and sludge compost (sewage sludge plus straw). Table 18.2 shows the increase in C in the 0–23 cm depth and the amount of organic matter added in each amendment in two periods. Doubling the rate of each amendment doubled the increase in soil C. Although appreciably more organic matter was added at the double rate of FYM compared to the single rate of sewage sludge both increased C in soil by the same

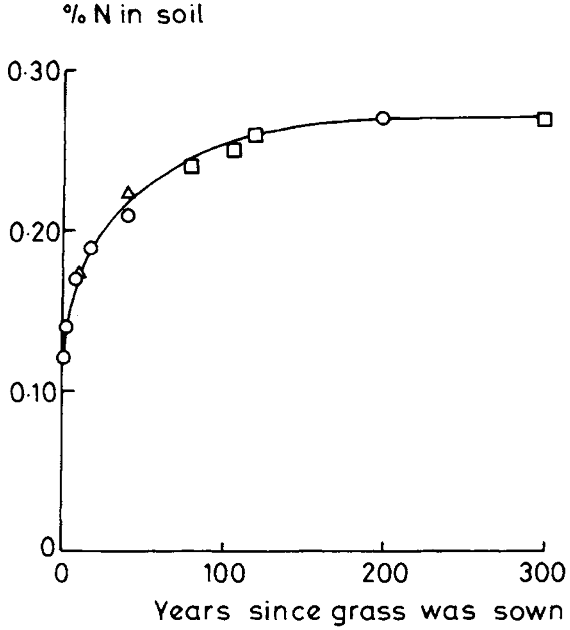

Fig. 18.3. Relationship between N content in unploughed grassland soil and year since grass was sown on a silty clay loam at Rothamsted. For further details see Johnston (1991).

amount. This is presumably related to the degree of decomposition of these amendments before they were added.

Sewage sludge and sludge compost were not applied after 1960, and FYM and FYM compost were not applied after 1967, so that SOM declined in all plots after

TABLE 18.2

Effect of FYM and sewage sludge, added at two rates, on the C concentration in the sandy loam soil of the Market Garden experiment at Woburn

Manure and annual application rate[a]	Period			
	1942–1950		1942–1960	
	Total addition t OM ha^{-1}	Increase in % C in soil	Total addition t OM ha^{-1}	Increase in % C in soil
FYM				
37.5 t ha^{-1}	50.2	0.45	102.2	0.50
75 t ha^{-1}	100.4	0.83	204.4	1.09
Sewage sludge				
37.5 t ha^{-1}	70.6	0.82	156.6	0.97
75 t ha^{-1}	141.2	1.38	313.2	1.84

[a]Fresh weight

these years. Although the decline started from different levels and at different times, the individual decay curves could be brought into coincidence by horizontal shifts (Fig. 18.4). This suggests that the readily decomposable SOM derived from each amendment had a similar composition. This would be expected if the material being decomposed was the end product of microbial activity rather than residual substrate.

The effect of different types of organic amendments on the formation of SOM was measured in an experiment started at Askov in 1956. The soil was a coarse sand subsoil (CS-subsoil) from the 50–100 cm depth on Sandmarken. Initially the soil had 0.3% C, 2.5% clay, and 2.4% silt. The subsoil was passed through a 4-cm mesh, mixed, and placed in concrete cylinders to a depth of 35 cm.

Every year 0.65 kg DM m^{-2} of dried FYM, mature cereal straw, sphagnum peat, or sawdust of known C and N content and widely different C-to-N ratios (Table 18.3) was mixed into the top 0–25 cm soil. Reference plots without organic amendments were also included.

The plots were cropped with a four-course rotation of spring barley, fibre flax, winter cereals (wheat or rye), and maize. Besides the organic treatments, maize received 23.4 g N m^{-2} in mineral fertilizer, and the other crops received 10.2 g N m^{-2} annually. Annual atmospheric deposition was taken as 1.0, 1.5, and

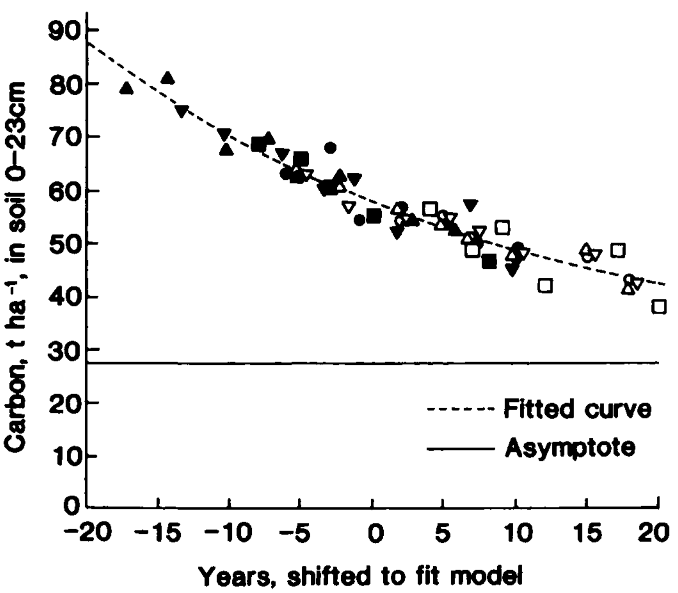

Fig. 18.4. Decline in C (t ha^{-1}) in soil of the Woburn Market Garden experiment. The decay curve for each treatment was shifted horizontally to fit a single decay curve. Treatments: FYM, single □, double ■; sewage sludge, single △, double ▲; FYM compost, single ○, double ●; sludge compost, single ▽, double ▼. Single (37.5 t ha^{-1}) and double (75 t ha^{-1}) fresh material each year during 1942–1960 for sewage sludge and sludge compost, and 1942–1967 for FYM and FYM compost (Johnston et al., 1989).

TABLE 18.3

Mean total C and N content in dry matter of the organic amendments given to the CS-subsoil at Askov during 1956–1986

Organic amendment	% C	% N	C-to-N ratio
FYM	35.2	2.05	17
Peat	49.9	1.13	44
Straw	46.0	0.64	72
Sawdust	48.7	0.15	325

2.0 g N m^{-2} during the periods 1956–1961, 1962–1969, and 1970–1986, respectively. All plots were limed occasionally to keep soil $pH_{(water)}$ in the range of 6.0 to 6.5.

The 0–25 cm layer of soil was sampled every four years, and Figure 18.5 shows the changes in C and N content during 1956–1987. Carbon increased from 0.27 to 0.57% and N from 0.019 to 0.046% in the reference plots without organic amendment. These increases are ascribed to organic inputs from stubble and rhizodeposits. Peat caused the largest increase in soil C, and the effects of FYM, straw, and sawdust were similar to each other. Soil N increased in the order: unamended < sawdust < straw and peat < FYM.

The observed increases in soil C and N in the reference soil were subtracted from those found in treated soils, and the accumulated input of C and N from each treatment (organic amendments, mineral fertilizer, and atmospheric deposition) was plotted against the C and N in the soil (Fig. 18.6). For FYM, straw, and sawdust the relationship between accumulated input (x) and soil concentration (y) of C and N could be described by a logarithmic regression equation: $y = a + b(\ln x)$ (Table 18.4). Although the effect of adding equivalent amounts of C in straw and sawdust appeared to be rather similar, there was more soil C with FYM. The r^2 value for soil C was, however, low for plots receiving sawdust. Extrapolating the accumulated C input to 23 and 32 kg C m^{-2} for FYM and straw, respectively (equivalent to 100 years of annual input), the C concentration in soil was calculated to reach 0.85 and 0.65% C, respectively.

The effect of peat additions appeared to be different and the relationship could best be described by $y = ae^{bx}$, where x = accumulated input and y = soil concentration of C and N, signifying an exponential increase in SOM with successive annual additions of peat. For soil sampled in 1987, the C-to-N ratio was 12 for the reference, FYM, and straw plots, but 18 and 21 for soil receiving sawdust and peat, respectively. Subtracting C and N contents in the reference soil allowed calculations of the C-to-N ratio of the SOM derived from the various organic amendments. SOM derived from FYM and straw had a C-to-N ratio of 11. In contrast the C-to-N ratio of SOM produced from sawdust and peat additions were about 28, suggesting that the availability of C to the decomposers was restricted in these strongly lignified substrates, and/or that deficiency in bioavailable N affected the decomposition processes.

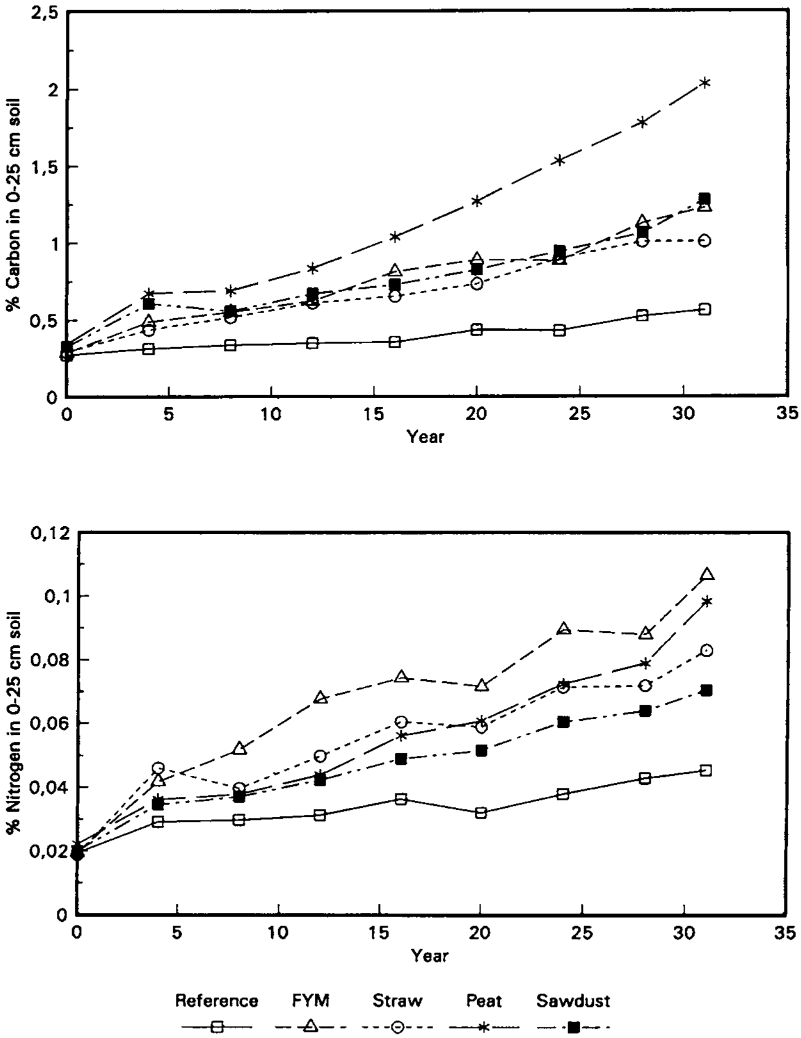

Fig. 18.5. The soil C and N content in the CS-subsoil at Askov after annual addition of 0.65 kg DM m^{-2} of FYM, cereal straw, sphagnum peat, or sawdust. The reference treatment received stubbles and rhizodeposits only.

The effect of above-ground crop-residue disposal was studied also in concrete cylinders as described above, but two soil types were used. One was the CS-subsoil, the other a sandy loam (termed SL-topsoil) taken from the 10–30 cm layer of a field with a long period in permanent grassland. The SL-topsoil had 9% clay and 13% silt and, in contrast to the CS-subsoil, a high initial carbon content (~3% C). The experiment had three treatments. In one treatment the soil was not fertilized and was permanently fallowed. The other two treatments had a four-course rotation of winter wheat, maize for silage, spring barley, and fibre flax; in one, all crop residues were

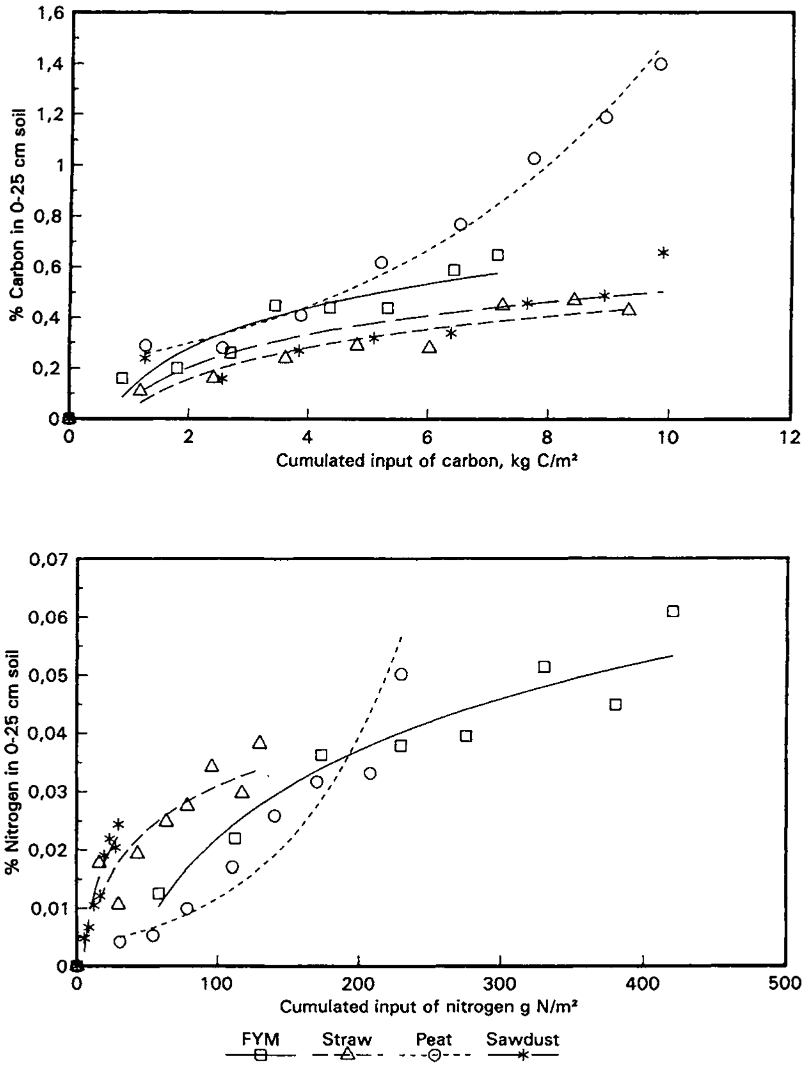

Fig. 18.6. The soil C and N content (contents in reference treatment subtracted) in the CS-subsoil at Askov as a function of cumulative input of C and N in FYM, cereal straw, sphagnum peat, or sawdust. Coefficients for the regression lines are given in Table 18.4.

incorporated into the soil, whereas in the other, only stubble and belowground residues were returned. Soil samples were taken every four years and analyzed for C and N content (Christensen, 1988a).

In the period 1956–1987 all treatments produced an increase in the SOM content of the CS-subsoil and a decrease in the SL-topsoil (Fig. 18.7 and Table 18.5). The return of all above-ground crop residues doubled the increase in the SOM content of the CS-subsoil compared to removal of residues, whereas the loss of SOM in the SL-

TABLE 18.4

Regression analyses on the relationship between accumulated C and N input (x) and soil content (y) of substrate-derived soil C and N in the 0–25 cm of the CS-subsoil at Askov (1956–1987)

Organic amendment	Regression equation	Carbon			Nitrogen		
		r^2	a	b	r^2	a	b
FYM	$y = a + b(\ln x)$	0.865	0.115	0.235	0.909	−0.0784	0.0218
Straw	$y = a + b(\ln x)$	0.869	0.032	0.180	0.752	−0.0191	0.0109
Sawdust	$y = a + b(\ln x)$	0.665	0.074	0.189	0.909	−0.0165	0.0115
Peat	$y = ae^{bx}$	0.977	0.198	0.204	0.939	0.00343	0.0122

topsoil was only half that found in soils to which above-ground residues were not added. Treatments providing the largest increase in SOM in the CS-subsoil were also those that prevented the largest decrease in the SOM rich SL-topsoil. The permanently fallowed plots on the CS-subsoil showed a small increase in SOM content over the years. This is ascribed to incomplete hand-weeding and the growth of algae.

This experiment demonstrates that in soils of low SOM and clay content used to grow annual crops, SOM contents can be raised significantly over a 30-year period,

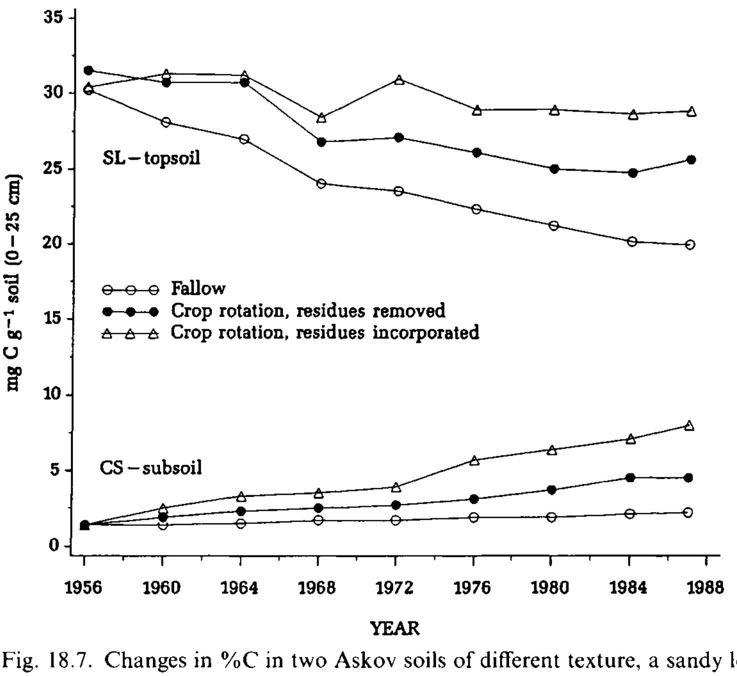

Fig. 18.7. Changes in %C in two Askov soils of different texture, a sandy loam (SL-topsoil) and a coarse sand (CS-subsoil) during 1956–1987. Samples were from the 0–25 cm of unfertilized and permanently fallowed plots and plots with a crop rotation of winter wheat, maize for silage, spring barley, and fibre flax. Aboveground crop residues were either removed or incorporated.

TABLE 18.5

Small-plot experiment on the effect of fallow and crop residue disposal on SOM content. Changes during 1956–1987 in the C and N content of 0–25 cm layer were calculated by linear regression of the type $y = a + bx$, where y was the % C or N in soil and x the number of years ($x_0 = 1956$). Soil was sampled every fourth year ($n = 9$) (Christensen, 1988a)

	Crop rotation with NPK fertilizer					
	Permanent fallow, no fertilizers		Crop residues removed		Crop residues incorporated	
	C	N	C	N	C	N
CS-subsoil						
Mean annual change (mg g^{-1} soil per year) (b)	+0.02	+0.003	+0.10	+0.007	+0.20	+0.014
Relative change over 30 yr (% of initial content)	+50	+62	+207	+147	+436	+307
SL-topsoil						
Mean annual change (mg g^{-1} soil per year) (b)	−0.33	−0.027	−0.23	−0.017	−0.09	−0.009
Relative change over 30 yr (% of initial content)	−34	−38	−22	−23	−8	−12

provided that inputs of crop residues are large. Removal of above-ground crop residues caused a slower increase in SOM levels (0.10 mg C g^{-1} soil yr^{-1}). Combining the results of this experiment with those from the experiment with different organic amendments suggests that an equilibrium level of about 1% C may eventually be reached in this coarse sand soil. The dramatic decrease in the C content of the SL-topsoil exposed to continuous bare fallow illustrates the quantitative importance of belowground crop residues in the SOM-forming processes and the deleterious effect on SOM of including bare fallow in the cropping system.

B. Effect of cropping system

In 1949 two Ley-arable experiments were started at Rothamsted, one on a soil with a long arable cropping history and 1.7% C, and one on a soil with 3.1% C that had long been in permanent grass. The yields of three arable "test" crops, which followed various three-year leys ("treatment" crops), were compared with the yields of the same test crops grown following three arable crops. Some plots of permanent grassland were retained in the experiment on the permanent grassland site, and some were sown to permanent grass on the old arable soil. In 1937 a similar experiment, with a three-year treatment phase but two-year test phase, had been started at Woburn, but unfortunately no continuous grass treatment was included.

Changes in SOM at Rothamsted (Fig. 18.8) can be summarized as follows: on the soil ploughed out from grass the SOM level declined steadily, but after 36 years it was still not as low as that in the old arable soil retained in arable cropping. The

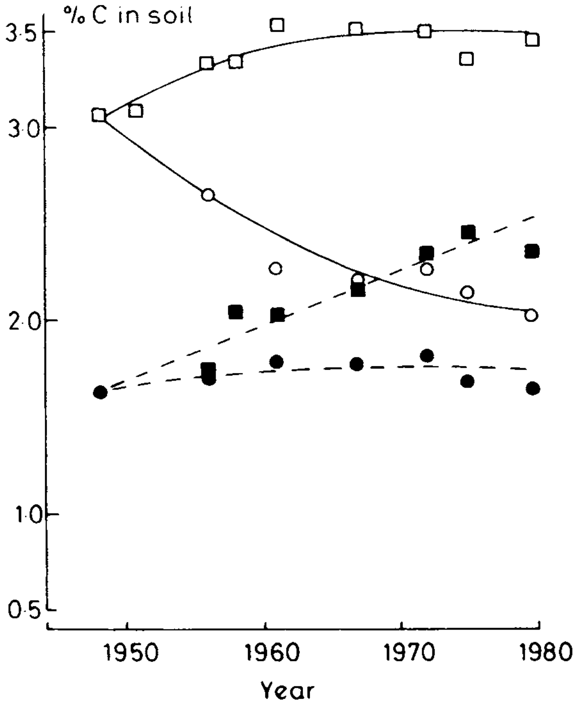

Fig. 18.8. Carbon content in Rothamsted Ley-arable experiment soils: Old grassland soil kept in grass (□), or ploughed and then growing arable crops continuously (○). Old arable soil kept in arable crops (●), or sown to grass and kept unploughed (■).

SOM content of the old grassland soil increased a little as a result of improved sward management, but in the old arable soil put down to permanent grass, the C content was still well below that in the old grassland soil even after 36 years. However, the changes in C with time under permanent grass in this experiment correspond well with the relationship between accumulation of SOM and time for other Rothamsted soils. In Figure 18.3 the 200- and 300-year values are from the Park Grass experiment located on a field that had been in grass for at least 200 years when the experiment started in 1856. The 300-year value was determined in 1959, 100 years after the start of the experiment. The relationship shows that it takes about 100 years for the equilibrium SOM content of this silty clay loam to change from that characteristic of an old arable soil to that of a grassland soil, and about 25 years to reach the halfway stage.

In both the Rothamsted Ley-arable experiments the effect of the three-year leys on total soil C was surprisingly small. Compared to the all-arable soils, the average increase in the C content in the 24th and 27th year after the experiment started was only about 10% for grass-clover and all-grass with N leys; the three-year lucerne ley did not increase soil C (Johnston, 1991).

Under all treatments the SOM content of the sandy loam at Woburn (Fig. 18.9) was less than the smallest amount in Rothamsted soil (Fig. 18.8), highlighting the

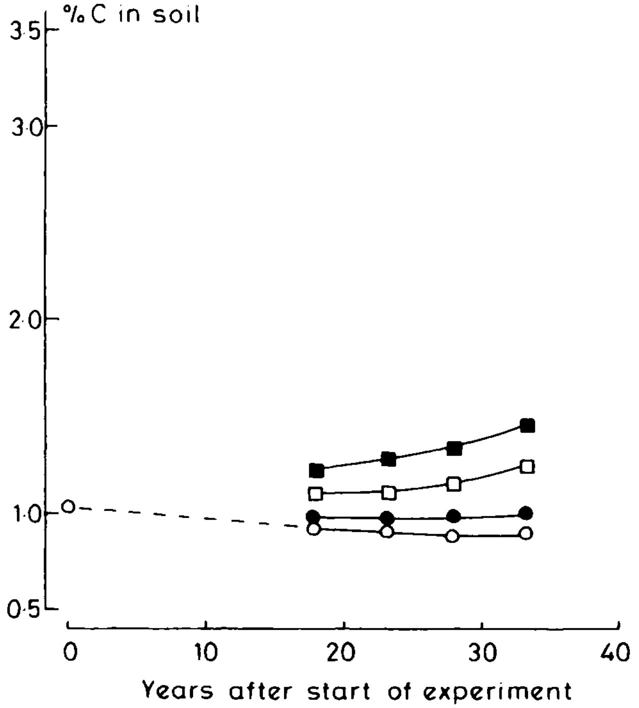

Fig. 18.9. Carbon content in Woburn Ley-arable experiment soil: All arable rotation with no FYM (○) or 37.5 t FYM ha^{-1} once in five years (●). Three years with grass ley and two years with arable crops given no FYM (□) or dressed with 37.5 t FYM ha^{-1} once in five years to an arable crop (■).

important role of clay in stabilizing SOM. The site chosen in 1937 for the Woburn experiment had been in a four-course arable rotation since 1876. Although the soil had only just over 1% C at the start, SOM continued to decline during the next 30 years under continuous arable cropping. However, yields have tended to increase provided that fertilizer applications, especially N, were adequate and cropping sequences were such as to minimize the build up of soil-borne pathogens. Other examples of changes in SOM with time have been given by Johnston (1991). In all cases the important observation is the slowness with which the SOM content changes under temperate conditions.

At Askov an experiment testing the effect of cropping system on the SOM content was started in 1956 on the SL-soil close to the B_3- and B_4-fields on Lermarken. All plots received nutrients corresponding to the 1 NPK dressing (see Table 18.1). Three crop rotations were tested: R1 (winter wheat, beets, spring barley, clover/grass); R2 (same as R1, but clover/grass was replaced by flax); R3 (same as R2, but beets were replaced by maize). The effects of these three rotations were compared with those of continuous fallow (Christensen, 1990). Changes in the C content of the plough layer (0–20 cm) during the period 1956 to 1986 are shown in Figure 18.10, and results of regression analyses on the C and N content are presented in Table 18.6. All cropping

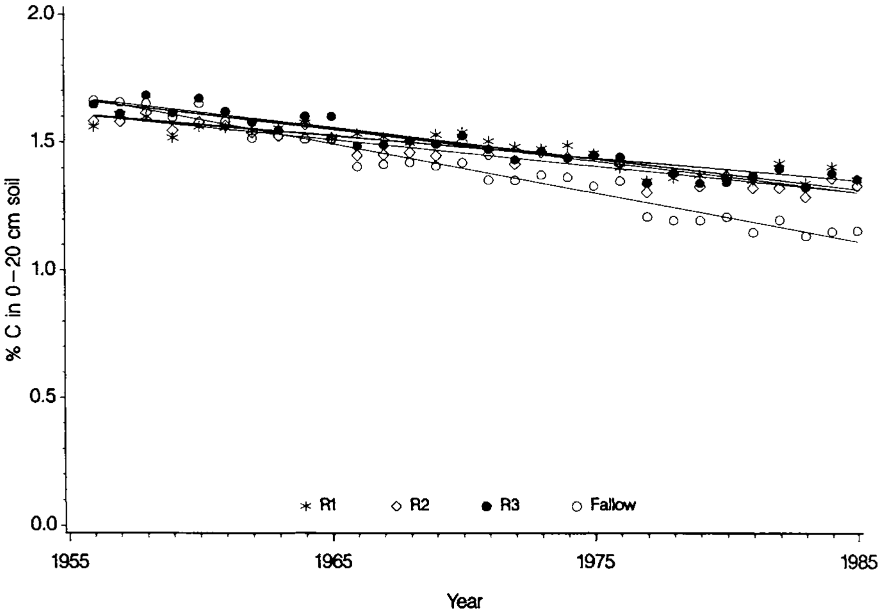

Fig. 18.10. The development in C content of soil under different cropping systems on Lermarken, Askov. R1 = rotation of winter wheat, beets, spring barley, and clover/grass; R2 = same as R1, but clover/grass replaced by flax; R3 = same as R2, but beets replaced by maize; Fallow = fallow every year. Lines indicate results of linear regression analyses (see Table 18.6).

systems caused the SOM content to decrease; the largest decline was observed under continuous fallow. Fallow soil lost 34% and 23% of the initial C and N content, respectively, over the 30-year period. Differences between the effects of the three crop

TABLE 18.6

Askov Lermarken: Effect of cropping system on soil organic matter content during 1956–1986, calculated by linear regression: $y = a + bx$, where y is C or N in soil and x is the number of years (x_0 = 1956). Soil from the 0–20 cm was sampled every year (n = 30)

Cropping system[x]	Mean annual decrease, mg g^{-1} soil		Relative loss over 30 years as % of initial value	
	C	N	C	N
Rotation 1 (R1)	0.088a[y]	0.0072a	16	15
Rotation 2 (R2)	0.102ab	0.0095b	19	20
Rotation 3 (R3)	0.118b	0.0107b	21	22
Fallow	0.190c	0.0109c	34	23

[x]R1 = winter wheat–beets–spring barley–clover/grass, R2 = same as R1, but clover/grass replaced by flax, and R3 = same as R2, but beets replaced by maize.
[y]Different letters signify significant differences (P ≤ 0.05).

rotations were small, although the inclusion of a clover/grass ley every fourth year appears to reduce the loss of SOM compared to rotations without ley.

IV. SOIL ORGANIC MATTER, CROP YIELDS, AND SOIL PROPERTIES

The previous section exemplified the effect of land use and soil management on the amount of SOM and its change over time. Soil quality is influenced not only by the amount of SOM in a given soil but also by SOM quality. The quality of SOM may be defined in terms of bioavailability of carbon to decomposer populations and mineralizability of organically bound plant nutrients, and by the chemical nature (composition, reactivity, and mobility) of SOM. Also, the occurrence of plant pathogens and the interaction among pathogens, their antagonists, and soil fauna may relate to the quality of SOM formed during turnover of different organic amendments. In this section we discuss the effect of SOM on crop yields and certain physical and chemical properties of differently fertilized soils.

A. Crop yields

The experiment on Barnfield at Rothamsted started in 1843, and P, K, Mg, and FYM treatments have remained unchanged. In 1968 four amounts of N were tested on potatoes, sugar beets, spring barley, and spring wheat, each grown three times in rotation between 1968 and 1973. Irrespective of the amount of N applied, yields of both root crops and spring barley but not of spring wheat were larger on soils with extra SOM resulting from applications of FYM since 1843 (Table 18.7). The best yield of barley was achieved with a much smaller amount of N on the FYM-treated soil than on fertilizer-treated soils.

These benefits from extra SOM became evident as crops with a larger yield potential were introduced and agrochemicals became available to protect this yield

TABLE 18.7

Yields (t ha^{-1}) of potatoes, sugar beets, spring barley, and spring wheat in 1968–73 on soils treated with fertilizers or FYM since 1843 on Barnfield, Rothamsted

Crop	Manuring	Fertilizer N applied[a]			
		0	1	2	3
Potatoes	FYM	24.2	38.4	44.0	44.0
tubers	PK	11.6	21.5	29.9	36.2
Sugar beets	FYM	27.4	43.5	48.6	49.6
roots	PK	15.8	27.0	39.0	45.6
Barley	FYM	4.18	5.40	5.16	5.08
grain	PK	1.85	3.74	4.83	4.92
Wheat	FYM	2.44	3.73	3.92	3.79
grain	PK	1.46	2.97	3.53	4.12

[a]N rates (kg ha^{-1}) 0, 1, 2, 3; to root crops: 0, 72, 144, 216; to cereals: 0, 48, 96, 144.

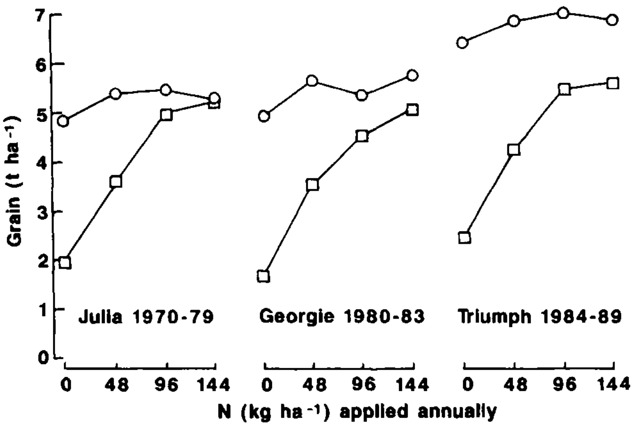

Fig. 18.11. Yields of three cultivars of spring barley grown on soils continuously manured with PK fertilizers (□) or farmyard manure (○) since 1852, Hoosfield Continuous Barley experiment, Rothamsted.

potential through the control of weeds, pests, and diseases. The benefit from extra SOM is also illustrated by yields of three cultivars of spring barley grown in three successive periods on the Hoosfield Continuous Barley experiment at Rothamsted (Fig. 18.11). In each period N was tested at four rates on soils with and without FYM since 1852. Although there was a large difference in soil C in 1975 (3.38 and 0.96% C with and without FYM, respectively), amounts of both readily soluble P and K were large and unlikely to restrict yields. Best yields of the cultivar (cv) Julia on both soils were not significantly different, but cv Georgie and then cv Triumph yielded more on soils with more SOM. For cv Triumph the yield difference was 1.2 t ha^{-1} grain.

SOM can affect the yield of arable crops through different mechanisms (e.g., release of nutrients, improved soil structure, and improved water-holding capacity), but these cannot be readily separated and quantified. The difficulty arises, in part, because in temperate climates it takes many years to establish soils with different levels of SOM and then to show that there are yield benefits from the extra SOM. Only then do researchers have the material and the incentive to do the appropriate research. Long-term experiments provide the material, and the recently observed differences in yield discussed above provide the motivation.

Trying to distinguish and separate the various factors through which effects of SOM are achieved must be a goal of future research. Recently attempts have been made to separate the effect of N from other possible, but not defined, factors, using yields for winter wheat grown on Broadbalk at Rothamsted (Johnston, 1987). Response curves were fitted to the yield data from fertilizer and FYM-treated soils during 1979–84. The form of the curve chosen was exponential plus linear, which gave in each case an optimum yield and its associated N application. Curves of constant shape were fitted to the data for each of the six years. Each year the shape was determined only by the response to the five levels of fertilizer N on PK-treated plots. This curve was then shifted until it fit the yields on the two FYM-treated soils, with and without fertilizer N, the horizontal and vertical shifts being determined separately. For both fertilizer and FYM treatments (i.e., low and high SOM contents), the maximum yield and associated N dressing could be determined. For each treatment the six response curves, one for each year, could be brought into coincidence by a diagonal shift to bring the maxima of each curve into coincidence. The combined curve for fertilizer- and FYM-treated plots indicated an average maximum grain yield of 6.94 and 8.33 t ha^{-1} with N dressings of 262 and 193 kg ha^{-1}, respectively. The overall fit accounted for 98.9% of the total variation.

The FYM and fertilizer curves could also be brought into coincidence by a diagonal shift (Fig. 18.12). The average horizontal component of this shift (−69.2 kg N ha^{-1}; range −39 to −109 for each of the six years) represents the N equivalent of the FYM-treated soils. The average vertical component of this shift (1.39 t ha^{-1}; range 0.1 to 2.3 for each of the six years) represents a unique benefit apart from that due to mineral fertilizer N applied as one dressing in spring.

A similar analysis of yield responses to different rates of mineral N fertilizer was applied to an experiment where clover-rich leys had been grown for one to six years

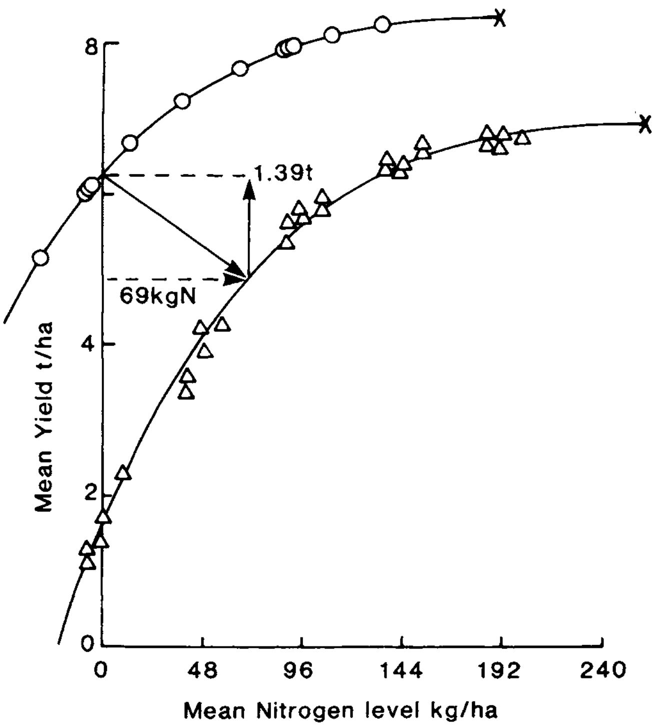

Fig. 18.12. Fitted response curves to grain yields of winter wheat 1979–84 on Broadbalk, Rothamsted, and the horizontal and vertical shifts required to bring the two curves into coincidence. Plots treated with mineral fertilizer (Δ) or FYM (O).

before being ploughed in. Short-term leys, especially clover-rich ones, can benefit the yield of arable crops that follow ploughing in of the leys, either from mineralised N or other beneficial effects of the added organic matter. Grass/clover leys were grown on the sandy loam at Woburn for one to six years before being ploughed under in the same autumn to prepare for a sequence of winter wheat, potatoes, and winter wheat crops. Following each ley mineral fertilizer N was tested at six rates on each crop.

Because the fertilizer N was applied in spring, the horizontal component of the diagonal shift required to bring the six curves into coincidence was attributed to an effective difference in N supply at this stage of growth, whereas the vertical component was attributed to factors other than spring applied N. Figure 18.13 shows for the first wheat the individual N response curves after the six leys and the response curve after they were brought into coincidence. Table 18.8 shows the horizontal and vertical components of the shift for all three crops relative to the one-year ley. For the first wheat the "nitrogen equivalent" of the ley, relative to a fertilizer N application in spring (the horizontal shift), ranged from 6 to 126 kg N ha^{-1}, and the "non-nitrogen" effect (the vertical shift) was approximately constant at 0.95 t grain ha^{-1} for the two- to five-year leys, but was less (0.40 t ha^{-1}) after the six-year ley. This may be explained by the fact that the maximum yield of wheat after the six-year ley was less than after the other leys, and yield hardly changed with increasing amounts of fertilizer N. This suggests that the crop was adversely affected by the amount of soil and fertilizer N available to it. For potatoes in 1988 the "nitrogen equivalent" of the two- to six-year leys was similar and very small, but the "non-nitrogen" effect ranged from 1.6 to 8.1 t tubers ha^{-1}, increasing

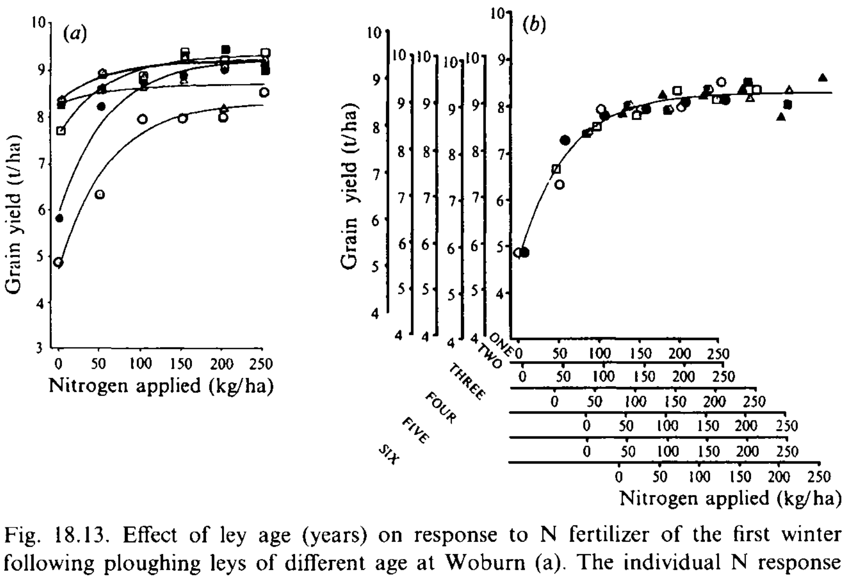

Fig. 18.13. Effect of ley age (years) on response to N fertilizer of the first winter wheat crop following ploughing leys of different age at Woburn (a). The individual N response curves were brought into coincidence by appropriate vertical and horizontal shifts (b). Ley age in years: 1 (○), 2 (●), 3 (□), 4 (■), 5 (△), and 6 (▲) (from Johnston et al., 1994).

TABLE 18.8

Vertical and horizontal shifts required to bring the yield/nitrogen response curves from Figure 18.13 into coincidence. Results from Johnston et al. (1994)

Ley age (years)	Wheat 1987		Potatoes 1988		Wheat 1989	
	Vertical shift[a]	Horizontal shift[b]	Vertical shift[a]	Horizontal shift[b]	Vertical shift[a]	Horizontal shift[b]
2	0.93	6	1.6	6	0.42	9
3	1.02	45	3.5	8	0.86	9
4	0.92	84	3.8	9	1.49	−2
5	0.90	86	8.1	−3	1.19	7
6	0.40	126	5.9	10	1.26	13

[a]Vertical shift is the estimated shift in maximum yield (t ha^{-1}) compared to first year ley of grain (1987 and 1989) or of tubers (1988).
[b]Horizontal shift is the estimated shift in effective N dressing (kg N ha^{-1}) compared to first year ley.

up to the five-year ley. In 1989, the third year after ploughing the leys, their "nitrogen equivalent" was again negligible, but the "non-nitrogen" effect ranged from 0.42 to 1.49 t grain ha^{-1}, increasing up to the four-year ley but not thereafter. This suggests that, even for winter wheat, SOM can play an important role in helping to achieve the yield potential of high-yielding cultivars.

B. Organic matter in soil size separates

Table 18.9 shows the C and N contents in clay-, silt-, and sand-sized primary organo-mineral complexes isolated from unmanured soils and from those given 1½ NPK and 1½ AM in the B_2-field on Lermarken (see Table 18.1). Clay associated SOM accounts for 57–60% of the soil C, and silt and sand hold 29–31% and 9–10%, respectively. The C-to-N ratio of clay SOM is lower, and that of silt higher, than the C-to-N ratio of the whole soil. Contents of C and N in clay, silt, and sand increased in the order: unmanured < 1½ NPK < 1½ AM. Although some trends are suggested by the data in Table 18.9 (e.g., in clay and whole soil C-to-N ratios), significant effects of the different fertilization regimes are not evident. If the SOM contents in clay and silt from unmanured plots are used as a baseline, then increases in C and N caused by applications of NPK and AM suggest that the AM- derived SOM contributes similarly to clay and silt SOM, whereas SOM derived solely from crop residues resulting from the use of NPK contributes relatively more to SOM in silt than to SOM in clay. Considering the greater stability of silt than of clay associated organic matter (Christensen, 1992), it may be speculated that in a soil in which similar total contents of organic matter have been reached either by the use of mineral fertilizer or animal manure, the manure-treated soil will contain a larger proportion of more labile organic matter than that of the fertilizer-treated counterpart.

These results suggest that the SOM produced during decomposition of the organic matter added with AM would preferentially end up in the clay-sized separate.

TABLE 18.9

Distribution of carbon and nitrogen across particle-size separates from unmanured, 1½ NPK, and 1½ AM plots in the B_2-field on Lermarken, Askov. Data from Christensen (1988b)

	Unmanured		1½ NPK		1½ AM	
	C	N	C	N	C	N
mg g^{-1} whole soil						
Clay (< 2 μm)	6.83	0.633	7.67	0.725	9.14	0.879
Silt (2–20 μm)	3.48	0.244	4.17	0.296	4.72	0.331
Sand (20–2000 μm)	1.10	n.d.	1.39	n.d.	1.43	n.d.
Relative distribution, %[a]						
Clay	58	67	57	63	60	66
Silt	29	26	31	26	31	25
Sand	9	n.d.	10	n.d.	9	n.d.
C-to-N ratio						
Clay	10.8		10.6		10.4	
Silt	14.4		14.1		14.3	
Whole soil	12.7		11.7		11.5	
Relative increase, %[b]						
Clay			12.3	14.5	33.8	38.9
Silt			19.8	21.3	35.6	35.7

n.d. = not determined.
[a]% of whole soil SOM.
[b]% increase above that in unmanured.

Whole soils and size separates from the Askov Long-Term Experiments on Animal Manure and Mineral Fertilizers and from the Broadbalk Continuous Wheat experiment were subjected to solid-state ^{13}C-NMR analysis to chemically characterize the nature of SOM in differently fertilized soils (Randall et al., 1995). The relative abundance of major functional SOM groups as calculated from NMR spectra is presented for whole soils in Table 18.10 and for clay and silt size separates in Figure 18.14. Aromatics, O-alkyls, and alkyls are the dominant functional groups in both the Askov and Rothamsted whole soils; N-alkyls contribute somewhat less. Acetals and carboxyls each account for about 10% of the spectral area.

Applications of animal manure and mineral fertilizers do not change the chemical nature of the NMR-visible SOM pool of Askov whole soils, the relative distribution of the different functional groups being similar for unmanured, 1½ NPK, and 1½ AM. For the Rothamsted whole soil samples, SOM in FYM-amended plots tends to be enriched in carboxyls and N-alkyls but depleted in acetals and alkyls compared to SOM in soil receiving N in mineral fertilizers (N_3PK plots). For O-alkyls and aromatics, there are no effects of long-term fertilization on the chemical composition of NMR-visible SOM.

For both soils, clay contains a smaller proportion of aromatics than silt-sized separates and, in the case of Askov, clay contains a larger proportion of alkyls. Silt-

TABLE 18.10

Relative abundance of major functional groups calculated from ^{13}C-NMR spectra taken on whole soils from Askov (unmanured, 0; mineral fertilizer, 1 ½ NPK; and animal manure, 1 ½ AM) and Rothamsted (Broadbalk: full mineral fertilizers, N_3PK; and farmyard manure, FYM) (Results from Randall et al., 1995)

Major functional group	Spectral region (ppm)	Askov			Rothamsted	
		0	1 ½ NPK	1 ½ AM	N_3PK	FYM
Alkyl	0–45	18.9	17.7	19.9	24.2	19.3
N-alkyl	45–65	13.9	14.7	14.5	12.4	16.6
O-alkyl	65–90	23.0	23.2	24.9	19.3	20.3
Acetal	90–110	13.6	12.6	11.7	15.2	9.5
Aromatics	110–160	25.0	25.2	23.2	25.6	27.3
Carboxyl	160–190	9.4	10.4	9.7	7.1	10.5

associated SOM has a higher proportion of acetals than clay SOM in Askov soils, but Rothamsted clay and silt do not differ in acetals. In contrast to Askov, the Rothamsted silt contains a much smaller proportion of alkyls than of O-alkyls. Thus, in contrast to whole soils, NMR results obtained from particle-size separates differ between soils.

The relative distribution of functional groups in silt associated SOM is little affected by fertilization regimes (Fig. 18.14). There is some tendency for animal manure to increase the proportion of N- and O-alkyls but decrease alkyls and carboxyls compared to the unmanured treatment for Askov but not for Rothamsted. SOM in clay-sized organomineral complexes appeared to change slightly more with differences in fertilization regime. For Askov clay, animal manure increases the proportion of alkyls and N-alkyl compared to the unmanured treatment, whereas the proportion of aromatics and carboxyls is reduced. Again, the reaction of the Rothamsted soil to fertilization is different from that of the Askov soil, the Rothamsted clay being enriched in aromatics and somewhat depleted in O-alkyls following annual dressings of farmyard manure. The overall conclusion for clay is, however, the same as for silt, namely that fertilization does not markedly alter the distribution of functional groups.

C. *Soil physical properties*

A range of soil physical properties has been examined in unmanured, 1 ½ NPK, and 1 ½ AM plots on Lermarken, Askov (Schjønning, 1985; Schjønning et al., 1994). The porosity of samples from the B_3- and B_5-field, taken in 1984 and 1985 respectively, is shown in Table 18.11. There is an increasing trend in soil porosity in the order: unmanured < mineral fertilizer < animal manure. Compared to 1 ½ NPK, 1 ½ AM increased the volume of < 30 μm pores, whereas the volume of larger pores was decreased. The associated increase in the amount of plant-available water

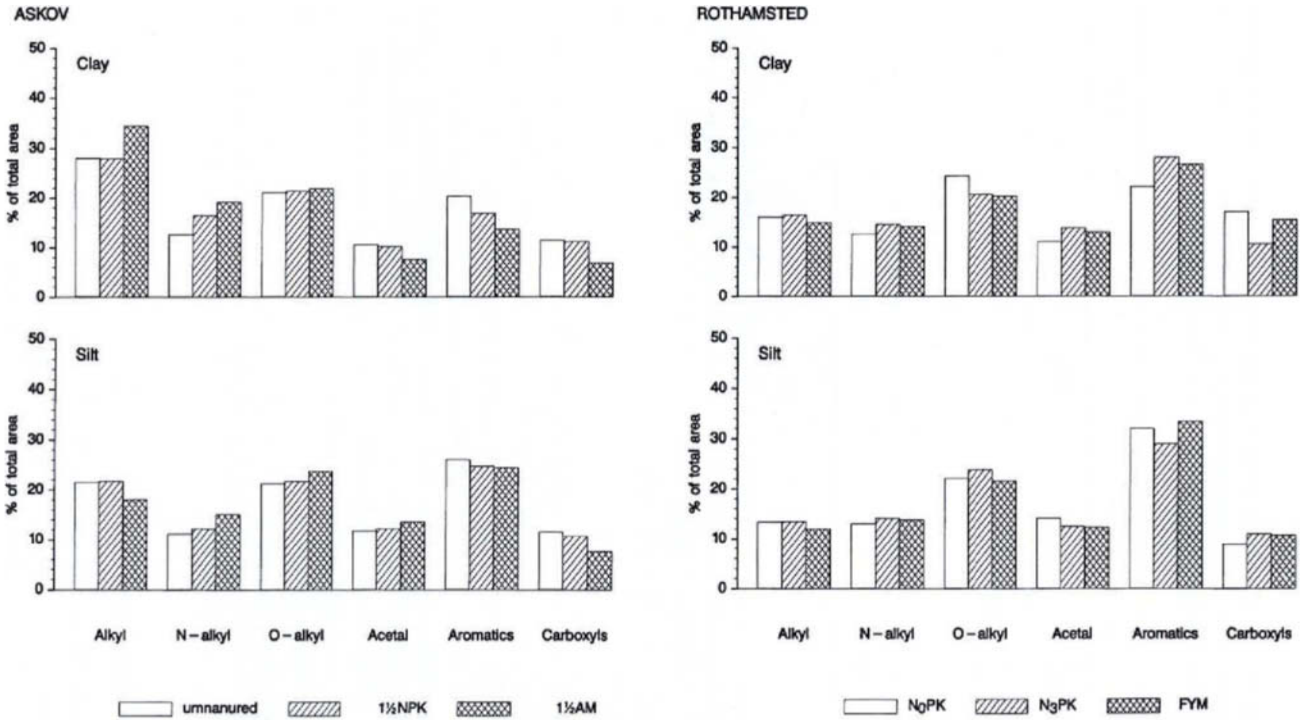

Fig. 18.14. Relative abundance of functional groups (as defined in Table 18.10) in SOM isolated with clay and silt sized organomineral separates from the B_2-field of Lermarken, Askov and the Broadbalk Continuous Wheat experiment, Rothamsted. Samples were from unmanured plots and plots receiving P and K (but no N) (N_0PK), full mineral fertilizer (1 ½ NPK and N_3PK) or animal manure (1 ½ AM and FYM) (data from Randall et al., 1995).

TABLE 18.11

Pore volumes (% of total sample volume) of unmanured, 1½ NPK, and 1½ AM soil samples from the 0–20 cm of B_3- and B_5-fields on Lermarken, Askov (Schjønning, 1985; Schjønning et al., 1994)

Soil depth and pore size class	Treatment		
	Unmanured	1½ NPK	1½ AM
B_3-field (sampled 1984)			
< 0.2 μm	8.2	8.1	8.7
0.2–30 μm	20.9	20.9	23.2
< 30 μm	29.1	29.0	31.9
> 30 μm	6.7	9.4	7.4
Total	35.8	38.4	39.3
B_5-field (sampled1985)			
< 0.2 μm	7.4	7.8	8.0
0.2–30 μm	17.7	18.5	19.7
< 30 μm	25.1	26.3	27.7
> 30 μm	12.9	13.6	12.6
Total	38.0	39.9	40.3

retained in 0.2–30 μm pores in the 0–20 cm layer corresponds however to only 3–4 mm water.

Samples from the B_3-field have also been analyzed for wet-stable macroaggregates (Table 18.12). Wet-stable aggregates in the range 2–8 mm were more abundant in fertilized plots than in unmanured plots, but for this physical soil property, the beneficial effect of the extra SOM formed through dressings of animal manure was distinct. The largest increase of 1½ AM over 1½ NPK was seen for the largest aggregates; 61% of the soil was in 6–8 mm aggregate under 1½ AM, whereas the corresponding values for 1½ NPK and unmanured were 43 and 32%, respectively.

TABLE 18.12

Wet-stable large aggregates in the 2–5 cm layer of unmanured plots and plots receiving mineral fertilizer (1½ NPK) or animal manure (1½ AM). B_3-field, Lermarken, Askov (Schjønning, 1985)

Aggregate stability	Unmanured	1½ NPK	1½ AM
Size class stability, %[a]			
6–8 mm	32	43	61
4–8 mm	40	48	64
2–8 mm	42	49	65
1–8 mm	49	56	71
Δ MWD, mm[b]	2.96	2.47	1.75

[a]Percent of aggregates in a given size class not broken down to < 1 mm after 5 min. of wet sieving.
[b]Δ MWD = change in mean-weight diameter of aggregate classes.

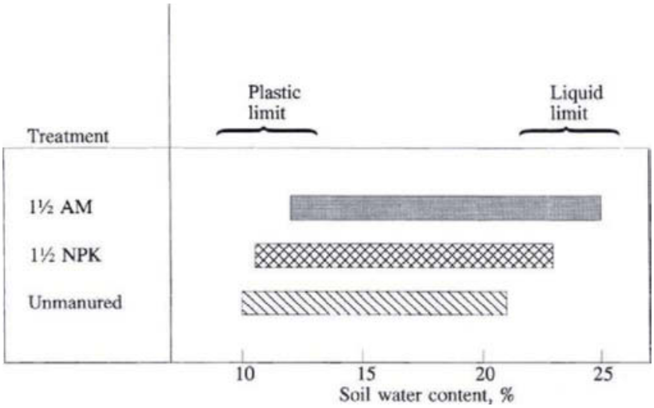

Fig. 18.15. Soil consistency limits for samples from the 0–20 cm of the B_5-field, Lermarken, Askov (data from Schjønning et al., 1994).

Also the stability of wet-stable aggregates was significantly improved following 1½ AM.

Soil consistency limits have been determined from drop-cone measurements on samples from the B_5-field (Schjønning et al., 1994). Plots receiving 1½ AM had plastic as well as liquid limits at higher soil-water contents than plots that were unmanured; plots with 1½ NPK were in an intermediate position (Fig. 18.15). The plastic limit signifies the water content at which the soil goes from a plastic to a friable state suitable for soil tillage. Thus for 1½ AM, the transition of the soil to a workable state occurs at a water content of 12.4% (w/w), whereas the water content of the unmanured soil has to be reduced to 9.8% before being workable.

Schjønning et al. (1994) also found that, despite the lower bulk density, soils taking mineral fertilizer and animal manure had a greater strength than unmanured soils at comparable water contents. Differentiation into cohesive and frictional soil strength components showed that unmanured soil increased steeply in internal friction upon drying, while the cohesion component remained constant. In contrast, soils receiving mineral fertilizer or animal manure increased in cohesion.

D. *Soil phosphorus*

Rubaek and Sibbesen (1995) determined a range of inorganic and organic P fractions in soils from the B_5-field on Lermarken. Because 1½ NPK and 1½ AM plots have received the same quantity of P over the years, the study allows both a quantitative and qualitative assessment of the P distribution in soils with different SOM contents. Plots receiving NPK and AM had similar amounts of total P, whereas unmanured plots were 40% lower in total P (Table 18.13). The proportion of total P present as organically bound P differed little between AM and NPK treatments (48 and 44%, respectively), whereas 62% of the total P in unmanured plots was organically bound. Values for plant available inorganic P were similar for

TABLE 18.13

Distribution of P among different chemically defined soil fractions from unmanured, mineral fertilized (1 ½ NPK), and animal manured (1 ½ AM) plots on B_5-field, Lermarken, Askov. (Results from Rubaek and Sibbesen, 1995)

P-fraction	mg P kg^{-1} soil			% of total-P		
	Unmanured	1 ½ NPK	1 ½ AM	Unmanured	1 ½ NPK	1 ½ AM
Total-P	318	553	535	100	100	100
Total-P_i	122	309	279	38	56	52
Total-P_o	196	244	256	62	44	48
Olsen-P	4.9	26	25	1.5	4.7	4.6
Mp-resin P_i	3.5	34	34	1.1	6.1	6.3
Hd-resin P_i	2.0	27	27	0.6	5.0	5.0
Hd-HCO_3 P_i	9	57	52	2.8	10	11
Hd-NaOH-P_i	55	162	141	17	29	26
Hd-HCl-P_i	26	46	38	8.1	8.4	7.2
Mp-resin P_o	0.6	2.5	2.8	0.19	0.45	0.52
Hd-HCO_3 P_o	20	23	29	6.3	4.2	5.4
Hd-NaOH-P_o	164	205	214	52	37	40
Hd-residual-P	66	80	84	21	15	16

Mp = macroporous resin method (Rubaek and Sibbesen, 1993).
Hd = Hedley fractions (Hedley et al., 1982).

1 ½ NPK and 1 ½ AM plots; these pools were much lower in unmanured soil. Although the various organic P fractions tended to be slightly higher following addition of animal manure, the overall conclusion is that the amounts and distribution of P forms are very similar for 1 ½ NPK and 1 ½ AM. In contrast, unmanured soil is significantly lower in P especially its inorganic fractions, and the relative distribution of the total P among the chemical fractions differs clearly from the manured plots except for Hd–HCl–P_i and Hd–HCO_3–P_0.

The response of arable crops to readily soluble soil P in the presence of adequate N and K was examined in an experiment at Rothamsted. The results (Table 18.14) show that there were large differences in yields between soils with different levels of organic matter when the soils were grouped by the level of soluble P they contained. This particular soil was among the most difficult to cultivate at Rothamsted, yet by many standards it was not intractable and yields were good. Extra soil organic matter probably had its effect through improved root exploration of the soil. This is the most likely explanation, because after the soil from each plot was sampled, air dried, and ground to pass a 2-mm sieve, it was mixed with half its weight of inert quartz to give a good rooting medium, and the mixture was cropped with ryegrass in pots in the glasshouse. Irrespective of soil organic matter content, when ryegrass yields were plotted against soluble P there was a single response curve, the shape of which was closely similar to that for the arable crops grown in the field on the soil of higher organic matter content.

TABLE 18.14

Yields (t ha^{-1}) of spring barley, potatoes, and sugar from sugar beets, on soils with different amounts of readily soluble P and at two levels of soil organic matter

Crop[a]	%C in soil	HCO_3-P, mg kg^{-1} soil				
		< 9	9–15	15–25	25–45	45–70
Barley grain	0.87	–	2.71	3.29	4.21	4.61
	1.40	3.18	4.78	5.20	5.00	5.46
Potato tubers	0.87	–	31.5	36.7	39.3	44.0
	1.40	26.6	43.0	45.3	46.4	47.8
Sugar	0.87	–	5.03	5.91	6.66	6.80
	1.40	2.45	5.92	7.06	6.73	6.85

[a]Barley grown in 1970, 1971; potatoes in 1971, 1972; sugar beets in 1970, 1972.

V. SUMMARY AND OUTLOOK

Any valid definition of arable soil quality inevitably incorporates some measures of SOM levels and dynamics, because SOM influences a wide range of soil features important to a sustained crop production. The SOM encompasses not only plant, animal, and microbial residues in all stages of decomposition, but also a diversity of heterogenous organic substances intimately associated with inorganic soil components. A number of abiotic factors exert control over SOM turnover, but for a given soil within a given climate, land use and soil management determine the amount of SOM.

In arable soils, management-induced changes in total SOM will be manifest and experimentally verifiable only over extended time periods, especially in temperate soils cultivated for centuries. Even when native and other permanently vegetated soils high in SOM are brought into cultivation or when long-term arable soils are converted to permanent grassland or woodlands, long periods are needed to follow changes in SOM. Therefore, long-term experimentation combined with simulation models are indispensable to determine subtle changes in the level of SOM. Recent examples of such combinations of SOM data from long-term experiments and SOM turnover models have been presented by Jenkinson et al. (1991, 1994), Paustian et al. (1992), and Parton and Rasmussen (1994).

The general decline in SOM content on the lighter-textured soils at Askov and Woburn is not an outstanding observation. Similar decreases have been reported for North American long-term experiments such as the Morrow Plots (Odell et al., 1984), the Lethbridge rotations (Monreal and Janzen, 1993), the Pendleton wheat–fallow experiment (Parton and Rasmussen, 1994) and the Sanborn Field (Wagner, 1982), and also for a Swedish long-term experiment on clay loam (Persson and Kirchmann, 1994). If these results are applicable more widely to present farming systems in the temperate region, implications for future soil fertility need to be evaluated in detail.

The results on generally declining levels of SOM substantiate the view that soils brought into cultivation recently may be significant contributors to the increased atmospheric CO_2 level, but, in addition, substantial amounts of CO_2 will already have been and continues to be released from arable soils that have been cultivated for centuries. Calculations based on results from the Askov long-term experiments suggest that the present annual net CO_2 emission from arable soils in Denmark could be as high as 2.7 Mt or equivalent to the CO_2 emission from the whole of the Danish national transport sector (Olesen, 1991). Jenkinson et al. (1991) have also discussed the possible effects of global warming on the release of CO_2 from soil.

Until recently crops grown with adequate mineral fertilizer have given economically acceptable yields, even though SOM levels have been maintained at levels only a little higher than in unmanured soils. Recent results suggest, however, that with modern high-yielding cultivars the yield potential and efficient use of fertilizer inputs may only be achieved on soils with more SOM. Through advanced plant-breeding techniques, genetic resistance to a number of plant pathogens has been introduced into modern crop cultivars. Combined with target-specific, flexible-dose pesticides and optimized soil management, modern cultivars are capable of giving large yields. There are indications, however, that these yields may only be realized on soils high in SOM, and that the effect of SOM cannot be fully replaced by increased levels of N fertilizer. In other words, the non-nitrogen effect of SOM on crop yields has become increasingly important for exploiting crop-yield potentials. Trying to distinguish and separate the various factors through which effects of SOM are achieved must be a goal of high priority in future research on soil quality. For organic farming systems relying heavily on on-farm recycling of plant nutrients, N_2-fixing leguminous crops, and exclusion of mineral fertilizers and pesticides, such research is even more urgently needed.

Animal manure, especially farmyard manure, has the potential of providing a significant increase in SOM contents, but it is unlikely that sufficient quantities are available for all soils growing arable crops. On sandy loams changes in SOM levels brought about by animal manure and fertilizer additions are relatively small. Differences between different arable crops in terms of SOM formation appear to be small compared to effects induced by crop-residue disposal and manure addition. Short-term leys included in arable crop sequences do little in terms of raising the SOM content. In contrast, permanent grassland and reversion of arable soil to woodland cause significant increases in SOM, but again it is important to recognize the slowness with which SOM levels change under temperate conditions in response to changes in land use.

The dramatic declines in SOM found in experiments with continuous bare fallow clearly demonstrate the importance of inputs of below-ground crop residues (roots and rhizodeposits) and stubble in SOM formation. The results also highlight the deleterious effect on SOM of including bare fallow in cropping systems, as also demonstrated in the long-term Lethbridge crop rotation study (Monreal and Janzen, 1993).

The physical properties of soil are intimately linked with SOM content. But because SOM levels change slowly, SOM effects on physical properties of soils from field experiments are rarely reported. In the light-textured Askov soil, the most distinct effect of the moderate SOM increase following applications of animal manure was on soil aggregation and aggregate stability, which both were increased substantially. The porosity, consistency, and strength parameters of the soil were also found to benefit from increased SOM levels. Similar data for Rothamsted and Woburn have been reported (Williams, 1978).

The results from the ^{13}C-NMR studies on soil samples from Askov and Rothamsted show that different fertilization (unmanured, mineral fertilizer, animal manure) produced only minor effects on the relative distribution of SOM functional groups in whole soils and clay- and silt-sized organomineral complexes. For both soils, however, clay was higher in alkyls and lower in aromatics compared to silt-sized separates. The results suggest that the chemical composition of the SOM accumulating in arable soils is dominated by microbial decomposition products and their interaction with the mineral matrix, and that various crop residues and animal manure cause the build-up of SOM of similar chemical nature. Similarly, the distribution of soil P across a range of chemically defined pools in the Askov soil was minimally affected by the form of P inputs. Compared to unmanured plots, however, addition of P either in animal manure or in mineral fertilizers was primarily reflected in the inorganic P fractions, in particular the plant-available pools.

Addition of strongly lignified substrates, such as sawdust and sphagnum peat, may produce SOM of a quality different from that of normal crop residues. It was found that the C-to-N ratio of the SOM formed from these materials was significantly higher than C-to-N ratios of FYM and straw derived SOM. Similar findings were reported for a Swedish long-term experiment testing different organic amendments (Persson and Kirchmann, 1994).

With this chapter we have provided examples of how SOM links to soil quality and how long-term or continuing field experiments provide a unique opportunity for research on the interactions between SOM, soil fertility, and agricultural productivity, but many lessons are still to be learned. The effect of SOM on the behaviour of microbial and faunal parameters is an area of considerable research potential. The potential of long-term experiments in elucidating the exchange of C between soils and the atmosphere will also be explored much more thoroughly in the future.

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GLOSSARY

Agroecosystem complexity: Classification of agricultural systems on the basis of biodiversity (i.e., number of different crops and other plants, and livestock and other animals), and the spatial (e.g., field size) and temporal (e.g., type and length of crop rotation) dimensions of the system.

Assessment of soil quality: The process of characterizing change in soil quality, by comparative (i.e., assessment of one system against another) or dynamic (i.e., continuously over time) means. The latter can involve the use of various approaches such as monitoring, statistics, and computer modelling.

Attribute of soil quality: Properties that reflect or characterize a soil process or processes that support a specific soil function.

Cybernetic systems: A system including an ecosystem, that is subject to regulation by feedback mechanisms arising from interactions among system components which eventually alters the original components; produces non-linear relationships between information inputs and system outputs.

Down-scaling: The reductionist approach by which an empirically defined system is dismantled to explain how it works according to its constituent parts and processes.

Dynamic soil quality: That aspect of soil quality relating to soil properties that are subject to change over relatively short time periods and that respond to management.

Ecosystems: Units of biological organization consisting of communities of organisms interacting with their physical environment.

Ecosystem health: The degree to which an ecosystem maintains its organization, function (i.e., energy and matter transformations), structure (i.e., information linkages, food webs, and biodiversity), and autonomy over time, and retains its resilience to stress.

Ecosystem management: An attempt to set explicit goals for ecosystems based on an understanding of ecological interactions and processes necessary to sustain ecosystem structure and function.

Ecosystem perspective: A broad view that soils and soil functions influence and are influenced by other ecosystem components and functions; encourages soil quality assessment on a large scale rather that at single points.

Extrinsic factors: Those factors apart from soil that influence plant growth, such as climate, topography, and hydrology.

Fertility erosion: The selective removal of portions of the soil organic matter and fine particles, rich in plant nutrients, by soil erosion processes.

Indicator: An indirect measure of a soil quality attribute that is a related or associated property (e.g., pedotransfer function, surrogate, or proxy). Indicators should be subject to standardization, specific to a quality attribute, sensitive to change, and easily measured or collected.

Inherent soil quality: That aspect of soil quality relating to a soil's natural composition and properties. Mainly includes static properties that show little change over time, such as mineralogy and particle size distribution.

Laboratory performance characteristics: Validation of soil quality methods by setting standards, especially by inter-laboratory comparison, for precision, limit of detection, measurement range and linearity, specificity and selectivity, and repeatability and reproducibility, etc.

Land quality: Quality of a broad land area or environmental system, including characteristics such as soil, water, climate, landscape (i.e., topographic and hydrologic factors), and vegetation.

Minimum data set: The smallest set of properties or attributes that can be used to characterize an aspect of soil quality.

Multifunctionality: Evaluating the quality or fitness of a soil for not one, but several functions.

Pedotransfer function: Functions that predict difficult-to-measure soil properties from readily obtained properties of the same soil. An interpolative technique based on regression equations. *Continuous* pedotransfer functions use soil properties as regressed variables. *Class* pedotransfer functions use soil type or horizon as the regressed variable.

Proxy property: See surrogate property.

Socio-economic dimension: A recognition of interactions among humans, their institutions (i.e., communities, market systems, agri-business, governments, religion), and natural ecosystems.

Soil biological quality: That aspect of soil quality derived from biological properties including living organisms and material (e.g. plants, microorganisms, meso- and macro-fauna, soil organic matter fractions) and processes associated with living organisms (e.g., C and N mineralization).

Soil chemical quality: That aspect of soil quality derived from chemical properties such as mineralogy, organic matter, pH, pE, electrical conductivity, exchangeable sodium, and cation exchange capacity.

Soil function: A role or task that soil performs such as a medium for plant growth, regulator or partitioner of water and energy, and an environmental buffer or filter; soil quality is defined by soil function.

Soil variability: The overall variability in soil properties arising from spatial, temporal, and analytical differences; an important consideration in soil quality evaluations.

Soil organic matter quality: That aspect of soil quality derived from the composition of organic matter, specifically the properties that characterize the labile or decomposable portion of organic matter.

Soil pedological quality: That aspect of soil quality derived mainly from inherent soil properties or indicators that reflect the processes related to the formation of soils, including soil spatial variability, soil landscape position and surface features.

Soil physical quality: That aspect of soil quality derived from physical properties such as soil pore space, soil strength, and structure.

Soil redistribution: The physical movement of soil components at the earth's surface, described by the following continuum of processes: detachment of soil, movement of soil (i.e., erosion), and deposition.

Soil resiliency: The capacity of a soil to recover its qualitative functions and dynamic properties, generally in a relatively short time frame, after some disturbance.

Soil quality: The concept of placing value on a soil related to a specific use. The seminal idea of categorizing the fitness of a soil for a certain use is now expanded to include ecological aspects, such as soil functions, that involve evaluating the capacity of a soil to function within specific ecosystem boundaries.

Soil quality control: The process of influencing or regulating soil management to sustain or improve the quality of soil.

Soil quality framework: An ecological approach that uses the sequence of function (i.e., what a soil does), process (i.e., those that support the function), properties (i.e., those critical to the process) or indicators, and methods to evaluate soil quality.

Soil quality monitoring program: The establishment and regular surveillance of monitoring sites or "benchmark" areas, to assess the magnitude and direction of change in soil properties over time. The process helps identify and track soil quality attributes.

Soil health: An approach to soil condition analogous to human or community health, by which the condition of a soil's properties and morphology are assessed against some optimum condition (i.e., soil-as-an-organism), or a soil's functions are assessed against the "goals" placed upon them (i.e., soil-as-a-community) or against an optimum functional state. Generally, soil health is synonymous with soil quality, except a soil may have poor inherent soil quality but still have good health.

Spatial statistical analysis: Statistical techniques used to analyse spatial patterns of soil and plant properties in ecosystems and agricultural landscapes.

Standardization process: The development by consensus of technical rules and protocols for the following steps in soil quality evaluation: site description, sampling, sample storage, sample preparation, analysis, interpretation of results, and presentation of results.

Statistical quality control: A method, analogous to the quality control of manufactured products, designed to help control and maintain soil quality processes within the range of natural variation. It involves experimental design, process monitoring and control, and ongoing adjustments of the system to maintain a process in an 'in-control' state.

Surrogate property (also **Proxy property**): Indirect properties or easily documented observations that can provide an appraisal or estimate of a difficult-to-measure property, or that can provide a substitute when specific data is missing or not available. Differs from pedotransfer functions in that the property is not obtained by regression analysis.

Tillage erosion: Displacement of soil due to tillage operations.

Topsoil: The surface soil layer moved or disturbed by cultivation, often enriched in organic matter. A common soil quality attribute or measurement, it always corresponds to at least the depth of the cultivated layer.

Up-scaling: The use of information at smaller scales to derive processes at larger scales. A challenging procedure since new interactions and processes can emerge in the latter, that were not present at the smaller scale.

REFERENCES INDEX