Global Issues in Water Policy 5

Andrew C. Chang
Deborah Brawer Silva *Editors*

Salinity and Drainage in San Joaquin Valley, California

Science, Technology, and Policy

Springer

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VOLUME 5

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ISSN 2211-0631
ISSN 2211-0658 (electronic)
ISBN 978-94-007-6850-5
ISBN 978-94-007-6851-2 (eBook)
DOI 10.1007/978-94-007-6851-2
Springer Dordrecht Heidelberg New York London

Printed on acid-free paper

Springer is part of Springer Science+Business Media (www.springer.com)

Kenneth K. Tanji
University of California, Davis, California
1931–2007

Preface

Salinity and drainage are challenges of irrigated agriculture in semiarid and arid climates that transcend history and geography. Three thousand years before Christ, the Mesopotamia culture emerged and prospered, aided by domestication of crops and animals and employment of a water conveyance and irrigation network. When the civilization crumbled millennia later, it was broad-spectrum socio-economical turmoil, poor governance, and institutional weakness that led to societal decline and eventually resulted in dilapidated water delivery infrastructure, rising shallow groundwater tables, salinization of soils, and crop failures. When and wherever irrigated agriculture has ascended in history, the populace sooner or later has been forced to cope with the threat of soil salinization. Examples today include Egypt, Jordan, China, Peru, India, Pakistan, Australia, and California. While water movement, salt buildup in soils, and plant injuries are molecular-scale processes governed by the natural laws, over time, it has been failures in public policy, institutions, and management, which have culminated in wholesale crises in irrigated agriculture.

At different temporal and spatial scales, salinity and drainage issues would take distinctively different forms and shapes. In 1983, the reproductive failures and deformed embryos of shore birds appeared at the Kesterson National Wildlife Refuge, a wetland fed by subsurface tile drain effluents from farmlands in the western San Joaquin Valley. A looming ecological catastrophe caused by irrigated agriculture was unfolding. At the process level, as drainage waters congregated in the terminus water body, micro-quantities of waterborne selenium bio-accumulated unnoticed through the aquatic food chain and quickly reached levels toxic to biota occupying the top ecological echelon. Instantaneously, the ecosystem harm brought the sustainability of irrigated agriculture in the Valley into question, a crisis at the regional level. Basin-wide options to address the issues were constrained by water rights, special interests, public policies and legal mandates, and political actions at the local, statewide, and federal levels. On farms, if irrigated agriculture were to continue, the irrigation and cultivation practices had to be adjusted to contain and/or eliminate the release of selenium via drainage water discharge. There were technology gaps to be closed. “How to fix it?” was a multifaceted dilemma.

In 1985, the University of California (UC), Division of Agriculture and Natural Resources, in response to the environmental crises, launched the UC Salinity Drainage Research Program, mobilizing resources and personnel for a concerted effort to tackle problems associated with salinity, selenium, and drainage in the western San Joaquin Valley. UC researchers initiated a diverse range of research studies, worked with local special service districts, and participated in inter-agency review panels. It was a period of focused academic pursuits by students and faculty alike and intense interactions with public agencies, water management professionals, and individual growers. Research findings were disseminated in presentations at workshops and conferences, in written reports distributed among concerned public agencies, in proposals and plans for feasibility assessments, and in technical articles in professional journals. It was the first time salinity drainage problems of an irrigated crop production region were systematically and comprehensively investigated.

The UC Center for Water Resources asked selected participants in the Salinity Drainage Research Program to critically recapture findings that over the time span of two-plus decades had been scattered in the scientific literature of diverse disciplines. As the subject matter is revisited and updated, we all benefit from the perspective of time and experience, additional research findings, new technologies, and greater knowledge base, to view the material with wisdom.

This compendium of 15 chapters is a collection of independent treatises, each depicting a distinctive salinity drainage topic with fresh perspective. As environmental scientists, engineers, and biologists, we are cognizant of the multiple scales at which complex issues should be examined, and we recognize the need to integrate those scales to recommend viable solutions. At first, the subject matter covered in this collection may appear random and lacking in relationship, but when the scales are invoked and integrated, then a mosaic of irrigated agriculture in time and space emerges. The following are synopses of the chapters:

Time, Geography, and Scale

Chapter 1 provides background information and a water distribution map to familiarize readers with the San Joaquin Valley, setting a stage for events and issues that developed.

Chapter 2 delineates the evolution of irrigated agriculture in the San Joaquin Valley over the temporal scale. Water rights doctrines, state and federal water policies, and infrastructure building played decisive roles in how water was distributed and used then and now. Processes that facilitated irrigated agriculture to blossom also led to delays in solving salinity and drainage problems. Later, the constraints were set forth by federal mandates in the Clean Water Act and Endangered Species Act and by eco-toxicological crises of selenium.

Chapter 3 depicts the geochemical and hydrological processes that define the San Joaquin Valley, including the physics, chemistry, and biology attributes that impact water management policies and strategies in the Valley.

Chapter 4 provides a comprehensive discussion of how scales entered into the salinity drainage research and management in the San Joaquin Valley. Scaling, the tools of integrating data obtained at different scales, is imperative in accurately assessing impacts at the regional level.

Biogeochemistry of Selenium

Chapter 5 elaborates chemical reactions that transform selenium when irrigation water passes through the soil profile and facilitates its transport to subsurface tile drains.

Chapter 6 explains the biochemical roles of plants in absorbing selenium from soils and transforming and volatilizing it as gaseous methyl selenium. The processes have the potential to reduce and eliminate selenium from the drainage water.

Chapter 7 describes how microbial reductive processes to precipitate selenium species found in the drainage water are affected by environmental factors and demonstrates a path to optimize the reduction of selenium.

Chapter 8 describes the aquatic chemistry and biology of selenium in evaporation ponds that are employed to retain and concentrate dissolved salts in the drainage water. Again, the processes may be employed to reduce the selenium load of the drainage water stream.

Coping with the Salts

Chapter 9 shows that evaporation ponds, while acting as the repository for selenium and dissolved salts in the drainage water, might be a threat to the safety of foraging shore birds. However, providing alternative and compensatory habitats for the birds can mitigate the potential hazards. The evaporation pond systems operated by the Tulare Lake Drainage District are a successful example.

Chapter 10 documents on-farm and plot-level irrigation provisions that would reduce agricultural drainage outputs and examines their effects on plant performance. These meso-scale practices might be implemented on a basin-wide level to enhance irrigation efficiencies and reduce drainage and salt disposal requirements.

Chapter 11 explores the on-farm drainage water reuse potential and tests the integrated farm drainage management (IFDM) approach. Drainage water may be retained on-farm by collecting, blending, and reusing it for irrigation of a more salt-tolerant crop. The secondary drainage water, in turn, may be reused on an even more salt-tolerant crop. The system, if properly scaled and operated, would reduce the drainage volume and concentrate the salts for final disposal.

Chapter 12 thoroughly delineates the technical merits of the reverse osmosis technologies in desalting agricultural drainage water, with emphasis on examining mechanisms of membrane clogging and on preventing microbial and mineral fouling of membranes and extending membrane life and operation time.

Chapter 13 shows how the wetlands in the San Joaquin Valley may be operated and through monitoring data evaluates their performances in accommodating the salt discharges from irrigated fields.

The Real Game: Public Policy and Management

Chapter 14 is a realistic policy analysis of the water management options for irrigated agriculture in the San Joaquin Valley and their respective outcomes, if implemented. In this exercise, the findings at the micro- and meso-scales are integrated for a basin-wide assessment.

Chapter 15 provides an international perspective on the sustainability of irrigated agriculture and returns to the thesis from the outset.

As editors, we remind readers that at different temporal and spatial scales, salinity drainage issues would take distinctively different forms and shapes.

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List of Figures

List of Tables

Chapter 1
Prologue

A quarter century ago, California unexpectedly had to grapple with an eco-toxicological crisis due to the bioaccumulation of selenium through the aquatic food chain of agricultural drainage water stored in the Kesterson Reservoir, which led to deformities in waterfowl embryos, reproductive failure, and sudden death in birds and fishes. The Kesterson Reservoir was a part of the National Wildlife Refuge in the San Joaquin Valley, the heartland of California's Central Valley where more than two million hectares (five million acres) of irrigated farmland are located. See water distribution map of Central Valley, California on the following page.

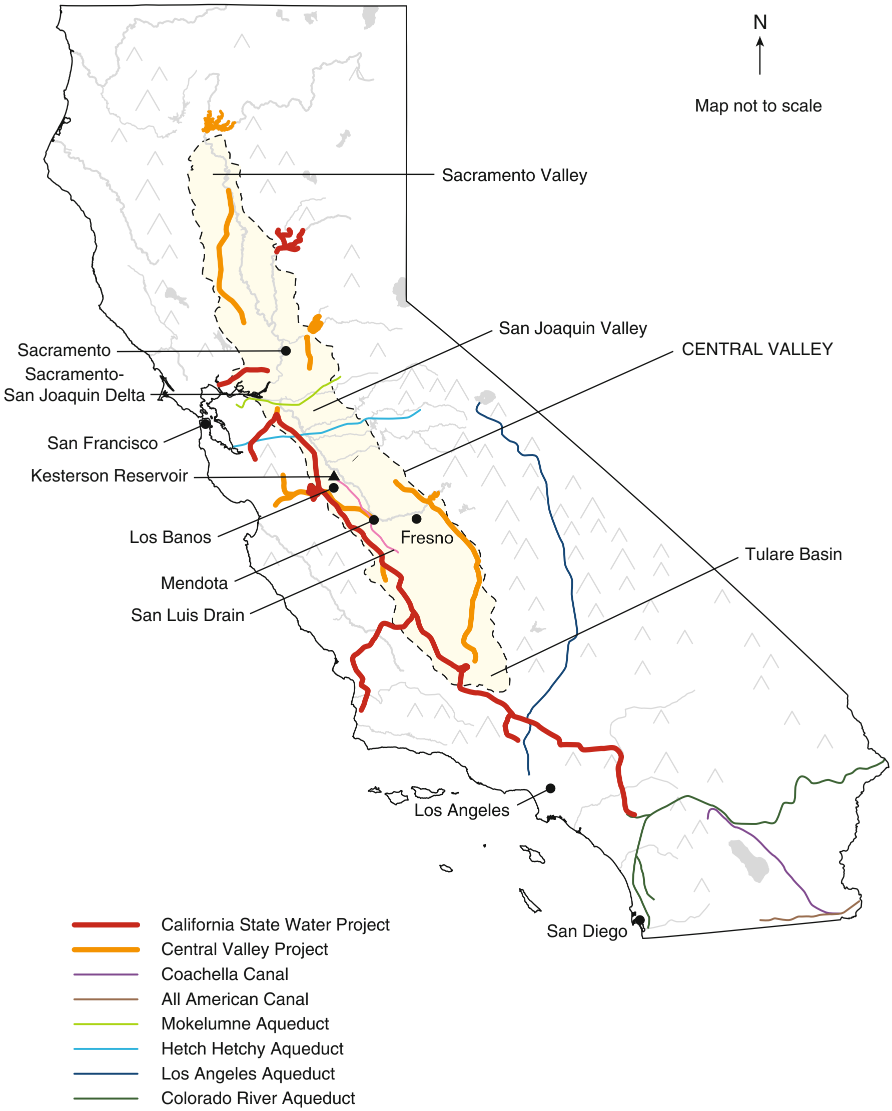
The environmental problem became so massive that scientists and policy makers doubted whether irrigation would be sustainable on the west side of the San Joaquin Valley south of Stockton. Lest the reader think scientists, farmers, and government officials should have been prepared in advance for this eventuality, it is important to know that environmental problems related to a toxic trace element, such as selenium contaminating saline drainage water from irrigated agriculture, had been unheard of worldwide; moreover, irrigation had been in practice for more than 100 years in the San Joaquin Valley without ever encountering selenium toxicity.

1.1 Irrigation Undergirds California's Leading Industry

Nature has blessed California with a remarkably long and sunny growing season and large areas of fertile soils. The weather is conducive to growing a wide range of crops, and in some areas two or three harvests are produced each year. The limiting input with respect to crop production is water. Hence, irrigation is a key feature of agriculture in California and a necessary input in arid areas of the state, such as the San Joaquin Valley.

Much of California is arid or semi-arid, and most rainfall occurs outside the April to November growing season. Mean annual precipitation ranges from 14.5 mm (0.6 in) in southern deserts to 1,285 mm (50 in) in the redwood forests of northern California. With virtually no rain during the growing season, much of California would be unfit for agriculture without irrigation. In fact, Mark Twin lamented,

A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_1,



Central Valley Water Distribution Map

"California has everything – usually at the wrong place." Twain's opinion of California is debatable, but "at the wrong place," and at the wrong time do aptly describe California rainfall. Providing water for irrigation has been a major feature of the growth of California agriculture from 1870 to the present. With irrigation, agriculture is one of California's major industries, generating more than $40 billion in revenue annually.

1.2 California's Central Valley: One Natural Drainage Outlet

California is known for its high-technology "Silicon Valley" with San Jose as its self-appointed capital, a region where companies, such as Adobe Systems, Apple Inc., ebay, Facebook, Google, Hewlett-Packard, Intel, Nvidia, Oracle, and Yahoo!, among many others, are headquartered, but, in fact, another California valley, the Central Valley, located east of the Silicon Valley and the San Francisco Bay area, ranks competitively in annual dollar value to the state. The Central Valley is a 650 km (400 miles) long bowl ranging from 63 to 97 km (40 to 60 miles) wide that is bounded along its entire length by the Sierra Nevada reaching over 4,400 m (14,490 ft) in elevation on the east side and the South Coast Ranges reaching 1,830 m (6004 ft) in elevation on the west. It spans 157,000 km^2 (60,480 mile^2) and is made up of three separate hydrologic regions: the Sacramento Valley, the San Joaquin Valley, and the hydrologically closed Tulare Lake Basin, as described below.

California's huge Central Valley, with its moderate winters and warm dry summers, has become a major agricultural production region characterized by many large-scale, commercial farming operations. Irrigation has transformed the Central Valley into the state's most productive agricultural region, a major producer of cotton, citrus, nuts, corn, grapes, vegetables, orchard fruits, grains and alfalfa. With allotments to environmental protection, agriculture, industry, and municipalities, California's available water resources are fully committed with little to spare.

1.2.1 Sacramento Valley

The most northerly region of the Central Valley is the Sacramento Valley (70,530 km^2 or 27,210 mile^2), which drains to the south. It is the location where the largest portion of the State's developed water supply originates. More water is exported from the Sacramento Valley via the delta of the Sacramento and San Joaquin Rivers, than from any other region in the state.

1.2.2 San Joaquin Valley

The more arid San Joaquin Valley (86,240 km^2 or 33,270 mile^2), includes the catchment basin of the San Joaquin River. The San Joaquin River flows from the

Sierra Nevada near Fresno in an east-to-west direction. Near the valley trough, close to the city of Mendota, the River makes an abrupt turn north and flows 160 km (100 miles) to converge with the Sacramento River to form the Sacramento-San Joaquin delta (referred to as 'Delta' throughout this volume), which drains through the Carquinez Strait into the San Francisco Bay. This is the only natural drainage outlet from the Central Valley to the Pacific Ocean, an important fact to know to appreciate the challenges faced by farmers, water managers, and state regulators and policy makers. The Kesterson National Wildlife Refuge, where selenium toxicity occurred, is located in the San Joaquin Valley.

Where the San Joaquin River exits the Sierra Nevada near Fresno, its flow is blocked by the Friant Dam and most of the flow in the River is diverted into the Friant-Kern canal, a primary source of irrigation water along the east side of the valley from Fresno to Bakersfield. Below the diversion, the San Joaquin River has little or no flow; however, along its route to the Delta, it receives flows from several large eastside tributaries draining the Sierra Nevada, including the Merced, Tuolumne, and Stanislaus Rivers. Flood flows from two smaller tributaries, the Fresno and Chowchilla Rivers, reach the San Joaquin River. Flows from the Calaveras, Mokelumne and Cosumnes Rivers do not flow into the San Joaquin River. Tributary in-flows from the drier coastal range on the west side of the Valley are very small.

1.2.3 Tulare Lake Basin

The Tulare Lake Basin, the south most subsection of the San Joaquin Valley with an area of 45,100 km^2 (17,390 mile^2), comprises the drainage area located south of the San Joaquin River drainage area. Major sources of surface water are the Kings, Kaweah, Tule and Kern Rivers, which originate in the Sierra Nevada and terminate within the basin. Only in extremely wet years does a small portion of the water flow into the Tulare Lake Basin drain north into the San Joaquin River. Imported water supplies are brought into the Tulare Lake Basin through a series of canals and aqueducts. These imported supplies make up more than one-half of the water supply used for irrigated agriculture in the Basin. Because the Tulare Lake Basin is essentially a closed hydrologic basin, all water supplies and the salts they contain remain in the basin and must be considered in all water management decisions.

1.3 The Master Drain That Never Was

> "What happened at Kesterson National Wildlife Refuge provides one more illustration of the long-known fact that irrigation projects without adequate outlets for drainage create unacceptable levels of salinity. The unexpected part of the scenario was that, given the right soils and geology, the process of drainage on irrigated lands can also concentrate trace elements to levels that can cause real harm to the biota" (National Research Council 1989).

During the years when major water projects were constructed in California, researchers, farmers, and public officials were cognizant of issues regarding soil salinity, water tables, and the need for drainage service in arid regions. In the 1970s and 1980s, state and federal officials and agency staffs were slowly making plans to provide long-term drainage service for irrigated agriculture in the San Joaquin Valley. The preferred plan was to collect drainage water in a drainage network, sometimes referred to as the San Joaquin Master Drain, serving farmland from southwestern Kern County (Buena Vista Lake Basin) to Gustine, and to discharge the drainage water into the Delta. The water would have been salty, but not nearly as salty as the Pacific Ocean, which would have served as the ultimate salt sink for the agricultural drainage water. The solution would not have been inexpensive, but it seemed technically feasible and permanent. Water tables would have been controlled and soil salinity would have been maintained within a range to support continuous production of a wide variety of highly valued crops, throughout one of America's most productive agricultural regions. Environmental permits would have been needed and a monitoring program would have been required for the discharge, but scientists and policy makers believed that obtaining these prerequisites could be accomplished, given sufficient political will and a substantial amount of public and private funding.

1.3.1 . . . And Then There Was Selenium

But while permits were being sought and political will was being cultivated, a new development arose that knocked the Valley-wide drainage solution off track so completely that it never has regained serious consideration. That development involved a single element on the Periodic Table, one that most observers in the San Joaquin Valley had never considered in any context. But from 1983 forward, everyone involved in irrigation and drainage in California has been quite familiar with selenium, which is needed in very small amounts to sustain healthy organisms, yet causes notable harm in slightly higher concentrations. According to the National Academy of Sciences, which publishes Dietary Reference Intakes, adult men and women need 55 $\mu g\ d^{-1}$ of selenium in their diets to avoid deficiency symptoms; however, consumption in excess of 400 $\mu g\ d^{-1}$ can lead to toxicity symptoms. The window of selenium concentration that supports health is narrow.

The planning for a Master Drain to carry agricultural drainage water to the Pacific Ocean, via the Delta, came to an abrupt end in the early 1980s with the discovery of selenium toxicities in fishes and birds in a set of drainage water holding ponds at the Kesterson National Wildlife Refuge. Selenium set the discourse of drainage service in California on a new path. A decades-long quest for an affordable, acceptable method of collecting, transporting, and disposing agricultural drainage water was derailed. Salt in the drainage water was not toxic, and it could be managed and blended. Nitrate could be removed using established methods. Selenium, on the other hand, caused toxicity, and the potential range of its impacts was not well understood at the time of its appearance at Kesterson National Wildlife Refuge.

The need for drainage service to sustain irrigated agriculture in the San Joaquin Valley had been apparent since the 1880s. Many researchers had studied the problem, focusing primarily on managing the salts and nitrogen in drainage water. Those constituents can degrade water quality and impair agricultural productivity, but they are not toxic to wildlife at levels observed in the Valley.

The same is not true for selenium, which can accumulate in the tissues of aquatic and terrestrial biota and cause significant harm as the selenium concentration increases.

The discovery of selenium toxicity in 1983 in the San Joaquin Valley was largely unanticipated. By May 1986, drainage water deliveries to the wildlife refuge had been terminated, along with any near-term plan to discharge agricultural drainage water generated by irrigated agriculture south of the San Joaquin River to the Delta. Since then, substantial public and private funds have been spent in search of a long-term solution to the Valley's drainage problem. An estimated $50 million was spent in sealing the drainage water holding ponds at Kesterson to prevent further selenium contamination. Additional funds were spent to plug the subsurface drains serving 17,000 ha (42,000 acres) in the Westlands Water District, a few miles south of Mendota, from which the drainage water had been collected.

1.4 Comprehensive "In-Valley" Disposal Solution: Still Pending

The Kesterson experience created great uncertainty regarding long-term drainage service for millions of acres of irrigated land on the west side of the San Joaquin Valley, including large portions of Fresno, Kings and Kern Counties. In 2007, the U.S. Bureau of Reclamation estimated the cost of constructing an out-of-Valley solution to serve just 153,781 ha (380,000 acres) in the Westlands Water District in Fresno County would be $2.7 billion. A similar sum would be needed for the alternative plan of removing selenium from drainage water and then disposing the treated wastewater within the Valley.

These estimated high costs for drainage solutions have motivated farmers and the staff members of irrigation and drainage districts to explore innovative strategies for reducing drainage water volume, managing salt loads, and removing salts and selenium from drainage water. Many districts have invested in research and development of new methods for actively managing salts and selenium on farms, due partly to the long-term perspective that disposal of drainage water outside the San Joaquin Valley will not be approved. Lacking such a disposal option, sustainability will require a cost-effective in-Valley disposal strategy.

A quarter century after the Kesterson crisis, a comprehensive in-Valley disposal plan is still pending; today, short-term agricultural productivity and profitability goals are weighed against other objectives that have become equally important to policy makers: long-term sustainability, protection of endangered species, and preservation of environmental quality. The 'greening' of California is impacting the available options for disposal of agricultural drainage water. The competition for limited water resources by growing urban communities in the state has also changed the landscape for irrigated agriculture. The challenges facing farmers, scientists, water district personnel, government regulators, water attorneys, and judicial adjudicators are not insurmountable, but formidable, requiring proactive leadership, sparing use of water resources, compromise, adoption and implementation of scientifically valid research results, and drainage planning by the whole community of stakeholders.

Chapter 2
E. W. Hilgard and the History of Irrigation in the San Joaquin Valley: Stunning Productivity, Slowly Undone by Inadequate Drainage

James D. Oster and Dennis Wichelns

> Between the Sierra Nevada and the mountains of the coast lies the great central valley, which is about 400 miles long and 62 miles broad. This central valley is a well-cultivated district, comprising nearly one-ninth of the entire State, and is almost throughout, where watered, of remarkable fertility.... For instance, the Plain of Fresno, which fifteen years ago was as bare as a threshing floor, is now one huge vineyard and fruit garden, intersected by thousands of irrigating channels, and full of fine country houses.
>
> (E.W. Hilgard 1893)

By the time University of California (UC) soil scientist E.W. Hilgard wrote these words in the first volume of *The Geographical Journal* (1893), he had traveled extensively in California's Central Valley and was impressed with its remarkable agricultural productivity made possible by irrigation. Depicted in the map in Chapter 1, the "great central valley" extolled by Professor Hilgard consists primarily of three hydrologic regions: (a) the Sacramento Valley north of Sacramento, (b) the San Joaquin Valley, the catchment basin of the San Joaquin River, and (c) the hydrologically closed Tulare Lake Basin, a subsection of the San Joaquin Valley. Irrigation demands and drainage needs are especially acute in the latter two sectors.

As part of his research activities, Professor Hilgard had visited with engineers from India, who described vast areas of formerly productive land in India that had succumbed to water logging and salinity within a few years after farmers began to irrigate, using water delivered by large surface irrigation schemes built to supplement or replace groundwater pumping. The cause of the problem was understood well by Hilgard that seepage from the elevated delivery canals and excessive application of irrigation water by farmers "relieved from the laborious processes

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_2,

of well irrigation" caused shallow groundwater to rise very near the soil surface, thus allowing salts to accumulate in the root zone (Hilgard 1886). The remedy was straightforward, although costly:

- Lower the elevated canals to reduce seepage and force farmers to lift water for irrigation, and
- Construct a regional drainage system to carry the saline subsoil water into rivers and the sea, "thus relieving the land more or less permanently of that scourge." (Hilgard 1886)

Because Hilgard, understood the similarity between agronomic conditions in India and California's Central Valley, he wrote passionately about the potential threat of water logging and salinity here, urging farmers and public officials in California to build regional drainage systems and to use irrigation water "sparingly" to prevent the otherwise inevitable, future harm from water logging and salinity that could occur to California agriculture:

> It is hardly necessary to go further into the details [of the problems occurring in India] to enforce the lesson and warning they convey to our irrigating communities. The evils now besetting the irrigation districts of northwest India are already becoming painfully apparent; and to expect them not to increase unless the proper remedies are applied is to hope that natural laws will be waived in favor of California. The natural conditions under which the irrigation canals of India have brought about the scourge, are exactly reproduced in the great valley of California; and what has happened in India will assuredly happen there also.
>
> (E. W. Hilgard 1886)

2.1 Professor Hilgard's Three-Pronged Prescription for Irrigation Management

A century before the selenium crisis at Kesterson, Hilgard recommended the following tripartite strategy for sustainable irrigation management, published in the *Bulletin of the University of California* College of Agriculture (Hilgard 1886):

- *Drainage Correlative with Irrigation.* Hilgard emphasized the necessity of developing drainage solutions concurrent with irrigation schemes.
- *Region-Based Solutions.* Hilgard wrote "[S]ingle individuals however, can do but little in the matter; the action to be taken must, of necessity, be that of whole communities."
- *Sparing Use of Water to Restrict the Rise of Alkali.* Hilgard warned that if irrigation was expanded without management strategies that used water sparingly to "restrict the rise of alkali" (as used by Hilgard, the term 'alkali' is synonymous with salinity), then water logging and salinity would plague California agriculture.

Hilgard's prescription for successful irrigation management is as valid in the twenty-first century as it was in the nineteenth! The current predicament in the

San Joaquin Valley is the outcome of failing to provide drainage at the time irrigation was developed, and failing to apply irrigation water efficiently from the outset. Hilgard was particularly prescient regarding the need for community efforts pertaining to drainage. Most farmers everywhere lack sufficient incentive to optimize their irrigation deliveries and manage deep percolation in ways that minimize off-farm impacts. Community involvement in the form of establishing regulations, providing incentives, and generating funds for infrastructure development is essential to prevent regional degradation of land and water resources. Natural laws have not been waived in California's favor in the past, and they continue to define irrigation opportunities and constraints, just as Hilgard warned 125 years ago.

To look forward, we must revisit what has occurred, in light of Hilgard's early warning, and examine the consequences to irrigated agriculture caused by the absence of a regional drainage system. We discuss why farmers, water agency managers, and public officials were reluctant to invest in a regional drainage system, a project Hilgard would have endorsed as essential for the sustainability of irrigated agriculture in California. We describe some of the reasons that the construction of a regional drain – an attempt by the whole community in California to provide for the necessity of drainage – was not completed. We also review some of the research and policies promulgated when it became clear that any long-term agricultural drainage strategy for the Valley must address not only water logging and salinity, but also selenium.

The presence of selenium in the drainage water generated by irrigation on the west side of the San Joaquin Valley makes the drainage problem, and community action needed to solve it, much more complex. The harm to wildlife due to elevated levels of selenium being transferred through the food chain weighs heavily against new efforts to develop a regional, out-of-Valley drainage solution. Many stakeholders believe the window of opportunity for conveying saline drainage water to the Pacific Ocean has passed; therefore, farmers and public officials must develop a sustainable, in-Valley strategy for maintaining agricultural productivity. Such a program likely will involve a smaller irrigated area, more careful irrigation water management – more "sparing" use of water, as Professor Hilgard recommended – and salt disposal within the irrigated areas along the west-side.

As this book goes to press, large areas of agricultural land have been retired on the west-side of the San Joaquin Valley. This action is due to an adverse combination of events: (a) chronic droughts in California that have reduced water supply; (b) court judgments underwriting land retirement through the purchase of irrigated lands by the federal government; (c) court judgments requiring reductions in surface water delivery to the Central Valley Project (CVP) and the State Water Project (SWP), both of which imported water to irrigate farmland in the Valley; and (d) the long-term problem resulting from the lack of a drainage outlet from the region.

2.2 History of Irrigation Development in California

The irrigation story in California actually predates Professor Hilgard's tenure at the UC in the 1880s. During the California Gold Rush period, when "forty-niners" (the name referred to those rushed for gold in 1849) who were seeking their fortune, converged on California and diverted water from the state's surface streams and rivers to assist their search. The infrastructures for water deliveries thereby evolved. The steady growth of irrigated agriculture in California was made possible by subsequent improvements in conveying networks, water management technology, supportive state and federal policy decisions that led to conversion of rain-fed lands to irrigated croplands, and the pursuit of financial returns in farming, food processing, and marketing.

Following initial Spanish colonization, agriculture in California gradually transitioned from a system characterized largely by cattle grazing and small farms to an industrial, highly mechanized, intensive agriculture based on irrigation. The transition reflected a consistent philosophical commitment to conversion of natural lands to croplands, and an increasing mechanization of farm practices. Figure 2.1 depicts both the total cropland under production in California from 1869 to 1997 and the steady growth of irrigated cropland during the same period. Tables 2.1, 2.2, 2.3, and 2.4 provides an historical, chronological outline of major developments in irrigated agriculture and drainage management in the San Joaquin Valley for the period 1840–2007, with a special emphasis on federal and state legislation enacted from the 1980s to 2007 to mitigate problems with selenium at the Kesterson National Wildlife Refuge.

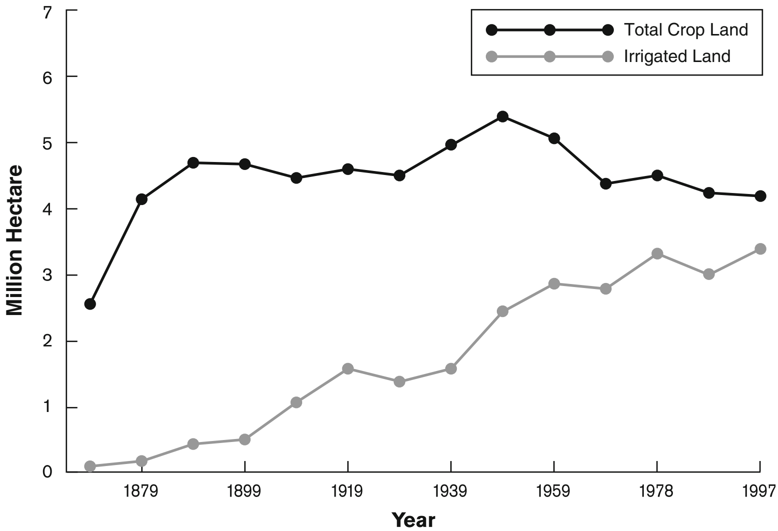

Fig. 2.1 Total and irrigated cropland of California, 1869–1997 (Johnston and McCalla 2004)

Table 2.1 Chronology of water development in San Joaquin Valley (SJV), Phase I 1840–1966: early expansion of irrigated agriculture and water supply

Event	Year	Importance for irrigation
Gold Discovered in California	1848	After gold was discovered at Sutter's Mill on the American River in 1848, thousands of new settlers flocked to California seeking fortune. Miners diverted water from the state's rivers and streams, searching for gold. After the Gold Rush (1848–1855), water conveying systems set up for mining were converted for irrigation of crops
Railroad Construction	1869–1875	Irrigation expanded as the railroads increased demand for produce grown by SJV farmers
Large-Scale Irrigation	1870–1880	By 1880, almost 81,000 ha were planted to cereals and alfalfa cultivated on irrigated croplands in the SJV
E.W. Hilgard, University of California Geologist, Proposed Drainage Solutions for Central Valley	1886	Professor Hilgard published his tripartite strategy for irrigation sustainability: "drainage correlative with irrigation, sparing use of water to restrict the rise of alkali (salts), and region-based solutions" in the *Bulletin of the University of California,* College of Agriculture
California Irrigation District Law (Wright Act)	1887	Passage of this state legislation allowed the creation of special irrigation districts in California. Farmers could organize for collective water development
Salinization of Irrigated Areas, SJV	1880–1890	Land on the east side of the SJV was forced out of production because of salinity and waterlogging
Federal Reclamation Act of 1902	1902	U.S. Congress passed federal legislation, creating the U.S. Bureau of Reclamation (USBR), authorizing federal construction of water projects in the western U.S., and subsidizing water for irrigation
California Water Commission Act	1914	California's system of administering appropriative water rights – issuing permits and licenses – was established by passage of the Water Commission Act
Deep Well Turbine Pump	1920	The invention of deep-well turbine pumps led to expansion of irrigated agriculture in the SJV because farmers were able to tap groundwater at depths up to 600 m, which supplemented their water supply from surface water diversions
California Constitution Amendment	1928	California Constitution was amended to require **"reasonable and beneficial use"** of the state's water resources (Article X, Section 2)
Central Valley Project (CVP)	1930s	The CVP is a federally funded water management program to expand irrigated agriculture in California by transferring water from water-rich northern California to the SJV. The CVP had no provision for salinity (salt) management

(continued)

Table 2.1 (continued)

Event	Year	Importance for irrigation
Land Subsidence	1920–1940	Excess groundwater extraction (overdraft) led to land subsidence in the SJV
Delta-Mendota Canal (CVP Expansion)	1951	The CVP began water deliveries to the SJV via the Delta-Mendota Canal
San Luis Drain	1955	Planning began for the San Luis Drain to the Delta[a]
San Luis Canal (CVP Expansion)	1960s	The CVP expanded surface water deliveries to the west side of Fresno County, permitting expanded growth of summer crops
State Water Project (SWP)	1960, ongoing	California voters approved the SWP in 1960, and it began operations in 1962. The SWP is the largest state-built water conveyance system in the U.S., providing a north-to-south transfer of water from northern California to irrigated agriculture in the SJV and urban areas of southern California
San Joaquin Master Drain (SJMD)	1965	The DWR[a] proposed the SJMD

[a] *Delta*, Sacramento-San Joaquin Delta; *DWR*, California Department of Water Resources

Table 2.2 Chronology of water resources developments in San Joaquin Valley (SJV), Phase II 1967–1982: continued expansion amid initial environmental concerns

Event	Year	Importance for irrigation
Nitrates in Agricultural Drainage Water	1967	The FWPCA[a] (predecessor of the U.S. Environmental Protection Agency) warned about the presence of nitrates in drainage water. Nitrate removal was proposed to address the proposed impact of the San Joaquin Master Drain (SJMD) on the Delta[a].
Drainage Plan	1968	The Central Valley Project (CVP) was required to obtain state and federal approval of its proposed Drainage Plan.
Porter-Cologne Water Quality Control Act	1969	This state legislation protects the quality of the state's water resources. Before Porter-Cologne, California did not have statewide regulations of the discharges of agricultural and industrial wastewater. Porter-Cologne established the State Water Resources Control Board (SWRCB) and nine regional boards charged with oversight, authority to issue permits and to set limits on the concentrations of toxic constituents in drainage water.
California Environmental Quality Act (CEQA)	1970	Permits to appropriate water are subject to compliance with the CEQA, except in emergencies.
San Luis Drain, First Stage	1971	The USBR[a] built a portion of the envisioned San Luis Drain between Five Points in Fresno County and Kesterson Reservoir.
Federal Clean Water Act	1972	Federal Water Pollution Control Act of 1972 is also known as the Clean Water Act. This landmark legislation enacted by Congress protects the quality of the nation's water supply by setting ambient water quality standards and effluent standards, preventing excess discharge of pollutants into the nation's streams and rivers.
Pollutant Discharge Permits	1972	The SWRCB administers the permits required for discharges of pollutants in agricultural drainage waters. Agriculture and irrigation operations lead to "non-point source" pollution in which one, specific source cannot be separated from the aggregate.
National Wildlife Refuge System Administration Act	1976	With passage of the National Wildlife Refuge System Administration Act, Kesterson Reservoir and the adjacent 1,870 ha (4,620 acres) became part of the Kesterson National Wildlife Refuge, a federally protected habitat for domestic shore birds and migratory birds of the Pacific flyway.
Kesterson Reservoir Water Supply (Fresh Water)	1972–1978	Kesterson Reservoir received freshwater inflows.

(continued)

Table 2.2 (continued)

Event	Year	Importance for irrigation
SJV Interagency Drainage Program (SJVIDP)	1975–1979	The SJVIDP prepared a drainage plan and proposed discharges near Chipps Island in the Delta[a].
San Luis Drain, Final Stage	1979	The planned, final stage of the San Luis Drain was never completed. The proposed discharge near Chipps Island was opposed by Bay Area interests (urban sector), due to its projected environmental impacts, and it was opposed by SJV farmers (agricultural sector) due to its cost.
Kesterson Reservoir Revised Water Supply	1980	Kesterson Reservoir began to receive subsurface drainage water from the Westlands Water District via the San Luis Drain, the portion of the San Joaquin Master Drain that had been constructed.
Kesterson Reservoir New Water Supply: Subsurface Drainage Water, Exclusively	1982	In 1982, all inflows into the Kesterson Reservoir consisted exclusively of agricultural subsurface drainage water.

[a]*Delta*, Sacramento-San Joaquin Delta; *FWPCA*, Federal Water Pollution Control Administration (predecessor of the U.S. Environmental Protection Agency [US EPA]); *USBR*, U.S. Bureau of Reclamation

Table 2.3 Chronology of water resources development in San Joaquin Valley (SJV), Phase III. 1983–1990: Selenium contamination at Kesterson Reservoir creates regional urgency

Event	Year	Importance for irrigation
Ecotoxicosis at Kesterson Reservoir	1983	Waterfowl deformities and reproductive failures at Kesterson Reservoir were documented by the USGS[a]. The DWR[a] withdrew its support for drainage water to be discharged to the Sacramento-San Joaquin Delta[a].
Crisis Reponses by Government Agencies	1984	The SJV Drainage Implementation Program (SJVDIP) was established. The SWRCB[a] oversaw regulation of water collection ponds at Kesterson Reservoir. The USGS[a] concluded that selenium contamination in the subsurface drainage water received at Kesterson Reservoir from the Westlands Water District was leached from the soil during irrigation. The soil parent materials are of marine origin.
SWRCB[a] Ordered Kesterson Cleanup Plan	1985	The SWRCB[a], a California state agency which oversees the state's water quality, ordered the USBR[a], a federal agency, to submit a Kesterson cleanup plan. (In 1979, Kesterson Reservoir had become part of the federal national wildlife refuge system.)
Kesterson Drainage Collection Ponds Closed	1985	The Secretary of the USDOI[a], which oversees the USBR, ordered the closure of drainage water collection ponds at the Kesterson National Wildlife Refuge in the SJV.
UC Salinity/Drainage Task Force	1985	The University of California (UC) responded to the Kesterson crisis with the appointment of the UC Salinity/Drainage Task Force, a group of research scientists representing several disciplines, who were funded by the State Legislature to study the problems at Kesterson and to coordinate with water and drainage district managers and local farmers to develop feasible solutions. One primary goal of the Task Force was to examine in-Valley solutions to the salinity, selenium, and drainage problems in the SJV.
Subsurface Drains at Westlands Water District Plugged	1986	Deliveries of drainage water to Kesterson Reservoir ceased. The Westlands Water District proposed and adopted a drainage management program to plug subsurface drains.
Drainage Service Facilities	1986	The U.S. District Court (federal court) ordered the US DOI to develop a plan for "Drainage Service Facilities" for the San Luis Unit of the CVP[a] by 1991.

(continued)

Table 2.3 (continued)

Event	Year	Importance for irrigation
Kesterson Reservoir Capped	1988	The SWRCB[a] issued an Order requiring Kesterson Reservoir to be capped with a 15-cm thick layer of clean soil to eliminate aquatic food chain exposure to selenium.
Shift in Public Perception and Policy Regarding Irrigated Agriculture	1983–1988	News coverage about the selenium contamination at Kesterson Reservoir influenced the urban public to become more concerned about the adverse environmental impacts of irrigated agriculture than its food and fiber productivity. And public policy makers required irrigated agriculture to comply with rigid interpretations of environmental guidelines.
DWR[a] Analysis	1988	Ron Robie, former Dierctor of DWR[a], characterized the failure to provide adequate drainage to serve the CVP[a] and the SWP as a tragic planning error.
National Research Council (NRC) Study	1989	The NRC wrote in 1989 that irrigation projects without adequate outlets for drainage "create unacceptable levels of salinity" and can also "concentrate trace elements to levels that can cause real harm to the biota."
Rainbow Report by the SJV Drainage Program (SJVDP)	1990	The "Rainbow Report" by the SJVDP outlined actions to manage salinity and drainage in the SJV.

[a]*CVP*, Central Valley Project; *Delta*, Sacramento-San Joaquin Delta; *DWR*, California Department of Water Resources; *SWRCB*, State Water Resources Control Board; *USBR*, U.S. Bureau of Reclamation; *US DOI*, U.S. Department of the Interior; *USGS* U.S. Geological Survey

Table 2.4 Chronology of water resources development in San Joaquin Valley (SJV), Phase IV. 1991–Present: search for sustainable solutions to salinity, selenium, and drainage

Event	Year	Importance for irrigation
USBR[a] Drainage Plan	1991	In December 1991, the USBR[a] published a plan to treat and dispose of drainage water in the SJV, entitled "San Luis Unit Drainage Program Plan Formulation Report and Draft Environmental Impact Statement." For details, see Johnston et al. 2011.
Federal Litigation Regarding Drainage	1991	A group of landowners in the Westlands Water District (WWD), known as the Sumner Peck plaintiffs, filed suit in federal court against the WWD and the US DOI[a] because of the lack of drainage, alleging damages. Their damages claim was not settled until 2002. (See below)
San Joaquin Valley Drainage Implementation Program (SJVDIP)	1991	The SJVDIP brought together University of California researchers, water and drainage district managers, and local farmers to work together to improve understanding of drainage issues and management options.
Central Valley Project Improvement Act (CVPIA, PL 102-575)	1992	Congress enacted the CVPIA to protect, restore, and enhance CVP management, particularly regarding fish and wildlife.
Grassland Coalition	1996	Farmers in the Grassland Watershed formed a regional coalition to manage drainage water volume and selenium (Se) loads. The source of Se in their Watershed is the soil itself. Irrigation mobilizes the Se that originates in the parent soil materials, which are of marine origin.
Waste Discharge Requirement for Selenium	1996	Regulatory agencies (SWRCB[a] and the Central Valley Regional Water Quality Control Board) established a new approach to manage selenium, specifying monthly and total annual load limits in specific water bodies. It was the first time a permit-type approach was used to regulate a non-point source (agricultural) discharge.
SJVDIP Evaluation	2000	The SJVDIP published its evaluation of the 1990 Drainage Management Plan for the Westside SJV in which it recommended strategies to minimize adverse impacts from salinity and drainage.
Federal Court Order to the USBR[a]: Provide Drainage to the San Luis Unit of the CVP	2000	On Dec. 18, 2000, the U.S. District Court (federal court) issued an amended Order, directing the Secretary of the US DOI[a], through the USBR[a], to "without delay, provide drainage to the San Luis Unit" of the CVP and to submit to the court a plan to provide drainage to the San Luis Unit.

(continued)

Table 2.4 (continued)

Event	Year	Importance for irrigation
Sumner Peck Litigation Settlement	2002	The Sumner Peck litigation was settled. The plaintiffs received in excess of $100 million from the federal government and the WWD. Approximately 14,973 ha (37,100 acres) were retired and non-irrigation covenants and drainage easements were placed on the lands.
Broadview Water District Land Retirement	2005	Landowners in the Broadview Water District, which was formed in the 1950s, sold their land to a neighboring water district in 2005 for the value of their water supply contract from the CVP, and discontinued farming. The stringent environmental requirements imposed by the SWRCB[a] for the permissible selenium concentration in drainage water and for the permissible total volume of drainage water made farming operations in Broadview unsustainable.
$2.7 Billion Cost Estimate by USBR[a] for Out-of-Valley Drainage Plan	2007 – Present	In 2007, the USBR[a] estimated an out-of-Valley drainage solution to the Delta for the agricultural drainage water from the WWD would cost $2.7 billion. The estimate included the cost of construction for drainage to the Delta, land retirement, drainage management, a reverse-osmosis (desalting) program, and a selenium treatment plant that would reduce a concentrated brine to <10 ppb. As of this writing, the USBR[a] is completing feasibility cost estimates, which are expected to confirm the need for reauthorization by the U.S. Congress to implement the plan.

[a]*SWRCB*, State Water Resources Control Board; *USBR*, U.S. Bureau of Reclamation; *US DOI*, U.S. Department of the Interior

2.2.1 Phase I. 1840–1966 (Table 2.1)

2.2.1.1 Expansion of Irrigated Agriculture and Expansion of the California Water Supply

Agriculture expanded rapidly following the California Gold Rush (January 1848–December 1855) and after California became the 31st state in the United States (September 1850). While gold mining continued, the gold fervor subsided and attentions were shifted to farming. The state's population grew quickly during this period, influenced, in part, by the Federal Homestead Act of 1862, which enabled settlers to gain ownership of quarter-sections (equivalent to 160 acres or 64.75 ha) of public land, if they would live on the land and improve it. The water conveying systems set up for mining were converted for irrigation water to support the agricultural enterprises of the new settlers in California.

The Wright Act, 1887

The California Irrigation District Law of 1887 (the Wright Act, Table 2.1) and its subsequent amendments, a milestone in California irrigation history, provided the legal mechanism through which farmers organized for collective water development statewide. The irrigated area cultivated in California doubled between 1869 and 1889, increasing from 2.4 to 4.8 million hectares (6 to 12 million acres) in just 20 years (Fig. 2.1). Today, water districts remain the fundamental infrastructure of distributing water throughout the state.

Miller and Lux Era

Henry Miller (1827–1916) and Charles Lux (1823–1887), two enterprising immigrants from Germany, acquired large tracts of riparian lands along the San Joaquin and Kings Rivers, and diverted water from these rivers to irrigate their alfalfa, cotton, and rice crops and to support their thriving livestock operation that grew to be one of the largest in America's history. Known as the "king of ranchers," Miller was a tycoon who owned more livestock and land than anyone else in the entire country, 0.57 million hectares (1.4 million acres), and he controlled ten times more through leases and grazing arrangements. Before the end of the nineteenth century, Miller owned a million head of cattle and 100,000 sheep, and his land holdings spanned 22,000 mile^2 (56,980 km^2) in three states (California, Oregon, and Nevada).

Miller and Lux undertook the development of significant irrigation projects. They built levees, irrigation ditches, three major canals, a reservoir in Kern County, and a dam across the San Joaquin River. The most significant water diversions occurred in 1872 when the San Joaquin River was diverted through their canal system west of Fresno (Johnston et al. 2011). With respect to irrigation and water management, Miller was a litigious businessman, who engaged in court battles to influence prevailing water rights laws. Miller believed in “riparian rights,” arguing that the owner of land along the banks of a water body should have the right to the full, un-diverted flow of that water; whereas, others held to the view that settlers downstream should have the right to divert water for legitimate needs, known as “appropriative rights.” Based on the long history of irrigation established in the late 1800s, many of the lands farmed by Miller and Lux more than a century ago retain “senior water rights,” as defined in the California Water Code (Sidebar 2.1).

By the early 1900s, two decades after Hilgard published his tripartite strategy for sustainable irrigation management, sizeable areas in the trough of the San Joaquin Valley (mostly on the east side) were being forced out of production because of salt accumulation (salinity) and high shallow water tables (water logging). As problems were surfacing in California at the turn of the century, federal policies were being enacted that fostered irrigation development in the western United States. President Theodore Roosevelt signed into law the Reclamation Act of 1902 (Public Law PL 57-161), which created the U.S. Bureau of Reclamation (USBR), declared the public goal of using water productively for irrigation in support of economic development in the 17 western states, subsidized water for irrigation, and authorized the construction of large-scale irrigation schemes in 17 western states, including California (Table 2.1). PL 57-161 provided farmers with interest-free loans on the construction portion of irrigation projects (GAO 1996), and farmers were not required to begin repaying the capital cost of a project until the project was deemed complete. In 1933, the Central Valley Project brought water that was regulated and stored in reservoirs in the Sacramento Valley to provide for municipal uses and irrigation in the San Joaquin Valley, a result of this 1902 federal legislation.

Sidebar 2.1 Surface Water Rights in California

Two types of surface water rights – riparian and appropriative – affect irrigation in California. The fact that California recognizes a dual system of water rights is sometimes called the California Doctrine. The laws and rights regulating surface water differ from the laws and rights regulating percolating groundwater in California.

Riparian Water Rights

A ***riparian water right*** is a “land-based” water right that derives from ownership of property adjacent to a surface source of water, such as a stream, river, or pond, or overlying a groundwater source, such as a subterranean stream. When California became the 31st state of the United States in 1850, it adopted the “riparian doctrine” of English common law, which entitled a landowner to use the surface water that flowed naturally by his or her

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Sidebar 2.1 (continued)

property. The property owner whose land abutted the banks of a body of water had a right to the undiminished, unaltered flow of the water because the property owner was viewed as owning the banks of the water source. This type of water right continues to be recognized in California. When a property is sold, the riparian water right remains with the property, transferring to the new owner of the land. Inherent in ownership of overlying land are riparian groundwater rights to subterranean streams.

A riparian water right does not require a permit, a license, or government approval; thus, the California Water Code does not have a statute defining riparian water rights. The concept of “reasonable and beneficial use” of water was added via amendment to the California Constitution in 1928 (Article X, Section 2). Prior to 1928, riparian water right holders were not required to put water to beneficial use, which led to legal disputes and conflicts with appropriative users of water (see below), who were always required to put water to reasonable, beneficial use. Irrigation is considered a beneficial use of water in California. Other beneficial uses include hydroelectric power generation, municipal and industrial uses, fish and wildlife protection, livestock watering, and recreation.

During drought, all riparian water right holders have equal priority. They are required to share the shortage equally, participating in a system of ***correlative rights*** (mutually related rights), with respect to surface water and groundwater. Conflicts between riparians are resolved on the basis of reasonable use, not on the basis of priority. Recourse to judicial determination occurs. In general, riparian water right holders have a higher priority collectively than appropriative water right holders. One of the exceptional circumstances in which a riparian water right holder must secure a permit occurs when the right holder plans to store water from one season to the next.

Appropriative Water Rights and Permits

California also recognizes ***appropriative water rights***, which are “use-based” water rights, as opposed to land-based riparian rights. An appropriative right is an entitlement to water based on the actual use of the water. A hierarchy of usage exists, such that the first user who diverts water has priority and the strongest rights (“senior rights”) when compared to a more recent, subsequent appropriator, who is said to have “junior rights.”

Appropriative water rights are rooted in the days of the California Gold Rush (1849) when “forty-niners” seeking fortune used and diverted water in the state’s rivers and streams to facilitate their search for gold. To stake their claims to water, miners would “post notice” to identify the water as belonging to them. The first miner to post a written claim had a prior right to the water. It was an ad hoc, self-governing “finders-keepers” system in which the first

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Sidebar 2.1 (continued)

miner to assert ownership received the senior right to the water. Other miners also had the right to divert available water from the same stream or river for beneficial use, but their rights were claimed in order of priority, with the "first-in-time, first-in-right" principle prevailing. To this day, the "first-in-time, first-in-right" principle is an important feature of modern appropriative water rights law in California, with "senior rights" extending to the earliest appropriator and "junior rights" extending to a more recent appropriator.

Until the early 1900s, no formal permission to use water in California was required from an administrative or judicial body. Then, in 1914, the California Water Commission Act created a statewide agency that had authority to issue and administer permits and licenses to appropriate the state's surface waters. Today, Section 1200 and subsequent sections of the California Water Code set forth water rights law in California, codifying appropriative rights. Section 1450 addresses the priority of appropriation. It states

> Any application properly made gives to the applicant a priority of right as of the date of the application until such application is approved or rejected. Such priority continues only so long as the provisions of law and the rules and regulations of the board are followed by the applicant.

The 'board' referred to in Section 1450 is the State Water Resources Control Board (SWRCB). The SWRCB issues appropriative water rights permits. The priority date of the permit determines the seniority of the appropriative water right. During water shortages, the most recent appropriator ("junior" right holder) must be the first to discontinue water use. Appropriators with "senior" rank can seek to enforce rights against other appropriators with lower rank, as long as the senior appropriator upholds the beneficial use requirement.

Note that the priority-based permit system for appropriative water rights in California exists in direct contrast to riparian water rights in which permits are not required. Moreover, the concept of "senior" and "junior" riparians is non-existent. Also unlike riparian rights, appropriative water rights may be sold or transferred, regardless of land ownership. Permits to appropriate water are subject to compliance with the California Environmental Quality Act (CEQA), except in cases of emergency when water transfers are exempt from CEQA review. Appropriative water rights can be limited by the "public trust doctrine," which holds that certain resources, such as water, are the property of everyone in the state (the public), and the state must hold these resources in trust for the public good. Navigable water bodies and their non-navigable tributaries cannot be diverted (appropriated) if it would harm their public trust value.

California Water Commission Act (1914)

The Water Commission Act established the state's current system of administering appropriative water rights. The Act created a statewide agency, the predecessor of the Water Right Division in State Water Resources Control Board (SWRCB), which had authority to issue and administer permits and licenses to appropriate the state's surface waters.

Groundwater Pumping Expands Irrigation

The invention of the deep-well electric turbine pumps during the 1920s was a major factor in the expansion of irrigated agriculture. Deep-well turbine pumps draw high-quality water for irrigation from deep aquifers, lying at depths of 600 m (2,000 ft) or more. Many farmers in the San Joaquin Valley adopted this technology to tap the groundwater at depths ranging from 100 to 600 m (330 to 1,970 ft) that supplemented their river diversions and to irrigate lands without access to surface water. Much of the land in the San Joaquin Valley that had been forced into retirement at the turn of the century lay idle until the 1920s when development of reliable electric turbine pumps and the energy to power them accelerated the expansion of irrigated agriculture by making available vast groundwater resources under the valley floor. Groundwater pumping lowered the water table in many areas, reducing the problems of water logging and salinity on the idle land (SWRCB 1977; Ogden 1988). Thus began the "conjunctive use" of surface water and groundwater for irrigation on the east side of the San Joaquin Valley, which is continuing to the present. In general, surface water that originates as runoff from the Sierra Nevada is naturally very low in salinity and is used to irrigate lands along the alluvial fans of the Valley, while drainage water from those lands recharges the groundwater and is used to irrigate lower lying areas.

Farmers on the west side of the San Joaquin Valley relied on a combination of river diversions and groundwater to support crop and livestock production (Sidebar 2.2). Many farmers began their irrigation projects using groundwater. By the 1930s, the negative effect of large-scale pumping began to manifest in the form of land subsidence over the more vigorously pumped regions along the west-side. Water was being mined from soft sediments, such as clays and silts, at rates far exceeding natural groundwater recharge. At some locations on the west-side, land subsided by as much as 8 m (26 ft) by the middle 1970s (Larson et al. 2001). Moreover, the declines in water levels as a result of excessive pumping added significantly to the cost of lifting water. As the depth to groundwater and the salinity of withdrawals increased, farmers began seeking political and financial support for a large-scale irrigation project that would bring surface water from northern California into the San Joaquin Valley. Both federal and state governments responded positively to their requests.

Sidebar 2.2 Groundwater Rights in California

Groundwater, by definition, is water located beneath the earth's surface in saturated zones. Subterranean groundwater streams in California and the underflow of surface streams are both subject to surface water laws in the state, which recognize two types of water rights, riparian and appropriative. The State Water Resources Control Board (SWRCB) issues permits to appropriate subterranean groundwater streams and the underflow of surface streams for beneficial and reasonable use, just as it issues permits to appropriate surface waters.

In contrast, the third legal category of groundwater in California, known as "percolating groundwater" – subsurface water that does not flow as an organized stream – is subject to distinct groundwater laws and treated differently from other types of groundwater. Percolating groundwater includes underground water basins and groundwater that has escaped from streams. Underground aquifers have the capacity to hold much more water than surface reservoirs. Cities, water districts, and other users can appropriate groundwater, diverting it over long distances from the groundwater basin of origin.

As of this writing, the method to appropriate percolating groundwater in California is simply to dig a well, pump the water, and put it to reasonable, beneficial use. No state permit is required to use percolating groundwater for irrigation or any other beneficial purpose in California. At present, California does not have a statewide management program or permit system to regulate appropriations of percolating groundwater. This lack of statewide regulation is a critical issue to know about California water rights. The state is not authorized by the California Water Code to manage groundwater basins. Owners of overlying land have "correlative rights" to extract as much groundwater as can be put to reasonable and beneficial use. Their rights are paramount to the rights of appropriators. More than a century ago (1903), the California Supreme Court decided that the "reasonable use" provision governing other water rights in the state also applied to groundwater usage.

In the absence of statewide regulations for percolating groundwater, some counties and existing groundwater districts in California have implemented regional regulations, including the passage of local ordinances, or the appointment of a "Water Master" to oversee local management of watersheds for the public good and to resolve disputes among appropriators, based on beneficial and reasonable use of the water.

Amendment to State of California Constitution (1928)

Article X, Section 2 of the California Constitution was amended in 1928 proclaiming "reasonable and beneficial use" of the state's water resources. Riparian water right holders from now on were required to put water to good use.

Central Valley Project (1930s)

The objective of the federal government to increase population settlement in the western region of the United States led to the formulation of several large-scale water resource development plans to expand irrigated agriculture in California. One of these projects, the Central Valley Project (CVP), was devised in the early 1930s to transfer water from water-rich northern California to the water-deficient San Joaquin Valley. The CVP now provides irrigation water along the west side of the Valley from Tracy in San Joaquin County to Kettleman City in Kings County. The CVP was developed pursuant to the provisions of the Reclamation Act of 1902. Water resource development by the federal government has had a profound influence on the economic, social, and political fabric of the San Joaquin Valley. The federal promise of affordably priced irrigation water supplies fostered a rapid expansion of irrigation on the west side of the San Joaquin Valley from about 1950 to 1970.

Delta-Mendota Canal (1951)

The Delta-Mendota Canal, one of the federally funded CVP canals, was completed in 1951. The opening of this canal provided a new, abundant, and low cost source of good quality water for irrigation. As indicated by its name, this canal carries surface water from northern California, via the delta formed by the Sacramento and San Joaquin Rivers, to a reservoir located near Mendota, a small town on the west side of the San Joaquin Valley. It serves several irrigation districts northwest of Mendota. In addition, the Canal carries replacement water for the San Joaquin River, pursuant to an exchange agreement between the federal government and several irrigation districts serving some of the original Miller and Lux lands along and near the river. In brief, the landowners in those districts have allowed the federal government to divert the San Joaquin River southward to provide irrigation water via the Friant-Kern Canal to farms along the east side of the Valley from Fresno to the southern end of the San Joaquin Valley. The exchange agreement provides farmers with a generous irrigation water supply, in comparison with other portions of the San Joaquin Valley. However, the water delivered from northern California contains more salts than water flowing naturally in the San Joaquin River from its source in the Sierra Nevada. As a result, the exchange agreement accelerated the pace at which salts were accumulating in soils on the west side of the San Joaquin Valley. On the other hand, water deliveries from the Delta-Mendota Canal helped arrest subsidence and contributed to the gradual recovery of piezometric heads in the aquifers toward pre-pumping levels. Although the heads (or pumping levels) in the confined aquifers have recovered significantly, only about 10 % of the total subsidence volume has been recovered, due to the plastic deformation properties of the sediments. The remaining 90 % constitutes groundwater storage that has been irreversibly lost. Similar subsidence problems occurred in other areas of the state, such as the Santa Clara Valley south of San Francisco Bay, and similar programs were developed to remedy it. The almost universal response to subsidence problems was to enhance water supply by building dams and canals to capture, store, and convey water to areas with substantial groundwater overdraft. From a political and economic perspective, irrigated agriculture was supported and public funds were provided to ensure the water supply needed to sustain it.

San Luis Canal

South of the area served by the Delta-Mendota Canal, a second CVP facility, the San Luis Canal, brought additional surface water to the western Fresno County near Mendota, beginning in the 1960s. Irrigated agriculture had expanded rapidly in this region of the San Joaquin Valley after World War II (Letey et al. 1986) based on pumping groundwater. Extensive pumping from beneath and above the Corcoran Clay Member, which lasted until the late 1960s, reached annual levels of about one million acre-feet (1.33 million ha-m or 1.33×10^{10} m^3), and lowered the water table by several hundred feet. The introduction of surface water in the western portion of Fresno and Kings Counties did not bring new land into production in that area. This imported water supply did, however, make it possible for the farmers to grow summer crops on all of the land, rather than have the summer crop production limited to the capacity of the available groundwater wells.

State Water Project (1960–1966 and Ongoing)

In addition to the federal government's policies and programs to enhance California's water supply, the state was also engaged in the process of plumbing California. The California State Water Project (SWP), approved by California voters in 1960, began operations in 1962. The SWP is the largest state-built water conveyance system in the United States. Supported by both urban and agricultural interests, the SWP was constructed on the same model as the CVP. Today, its facilities consist of pumping and power plants, reservoirs, lakes, storage tanks, canals, tunnels, and pipelines that capture, store, and convey water to more than 25 water agencies, known as SWP contractors, which receive annual allocations of water based on contracts that will expire in 2035. From its inception, the SWP has provided a north-to-south water transfer: Northern California water is supplied to both urban areas of southern California and irrigated agriculture in the Central Valley to the southern portion of the Tulare Lake Basin. Irrigation and water districts served by the SWP and the CVP have similar irrigation development profiles. A part of the SWP, a large reservoir known as the Oroville Dam was built to capture runoff from the Sierra Nevada (the Feather River Basin), and a new pumping plant was built to take water from the Sacramento-San Joaquin river delta (the Delta) and deliver it southward via the California Aqueduct. A CVP/SWP regulating reservoir (San Luis Reservoir) was constructed south of the Delta.

The San Luis Unit of the federal CVP and the SWP, each authorized in 1960, began delivering Northern California water to agricultural lands in the southern San Joaquin Valley in 1968. Together they were to provide water to irrigate about 405,000 ha (1,000,000 acres). By 1970, there was a reliable and abundant water supply for irrigated agriculture in the San Joaquin Valley, and the amount of irrigated land was again increasing (Fig. 2.1).

Completion of the CVP and the SWP increased water storage in California by about 16.8 million acre-feet or about 21 km^3 (Johnston and McCalla 2004; DWR 2009). The average annual combined yield of the two projects is about 10 million acre-feet (12 km^3). The increased water available for irrigation was adequate to irrigate an additional 4 million acres (1.62 million hectares). Thus, from 1940 through 1970, the total irrigated cropland in California increased from 4.2 million acres (1.7 million hectares) to 7.4 million acres (3.0 million hectares), representing about two-thirds of California's crop production land (Johnston and McCalla 2004). By 1978, irrigated agriculture accounted for about 8.5 million acres (3.4 million hectares), or about 74 % of California cropland (Fig. 2.1). This substantial increase in irrigated land was mostly in the SWP service area in Southern Kings and Western Kern Counties where groundwater had not previously been available. Currently, about two thirds of the average annual yield of the CVP and SWP is delivered to agriculture (DWR 2009).

Initial Salinity and Drainage Management Plans, 1950s–1960s

Farmers, who were strongly advocating the expansion of the CVP to irrigate lands on the west side of the Valley located within the Tulare Lake Basin, knew that water tables would rise once surface water was imported from northern California and they decreased their use of groundwater. The reasons why water logging and salinity occurred at the turn of the twentieth century along the east side of the San Joaquin Valley were well known and understood in the farming community. Efforts to include drainage service in the federal San Luis Authorizing Act of 1960, (PL 86-488) were successful. This inclusion was unique. Drainage service for salt management was not included in any of the early development plans for importing water from outside the San Joaquin Valley

In the following six paragraphs, we quote Johnston et al. (2011):

"As late as 1949, the U. S. Department of the Interior, Bureau of Reclamation (USBR)'s Comprehensive Report on Planned Water Resources Development in the Central Valley (USBR 1949), made no mention of salt management (SWRCB 1977). The only official reference to the problem was contained in these 1946 comments by the U.S. Department of Agriculture on the draft report, which stated:

> The plan does not discuss drainage or include costs relative to constructing or operating drainage systems. In the light of experience with lands that have been irrigated, we feel that properly integrated plans for drainage should be made a part of any proposed new irrigation development plans (SWRCB 1977).

"The first consideration of salt management occurred in the 1950s and 1960s as large-scale water development plans begun to take shape and salt management plan begun to be incorporated into the process. The 1955 Feasibility Report for the San Luis Unit of the Federal Central Valley Project, in the west-central part of the valley, recognized the need for drainage and proposed an interceptor drain for the Unit (USBR 1955). In addition, a California Department of Water Resources (CDWR) report to the California Legislature recommended the state study a 'comprehensive master drainage works system' for the valley. The California State Water Plan prepared by CDWR included the concept of a "valley-wide master drain" and, in looking at the southern portion of the valley, assessed the problem in the 1957 by stating,

> Drainage must be considered an integral and indispensable part of the development and utilization of water resources. Adequate provision must be made, therefore, in the total program (DWR 1957).

"The San Luis Unit of the Federal Central Valley Project (CVP) and the State Water Project, each authorized in 1960, begun delivering northern California water to agricultural lands in the southern San Joaquin Valley in 1968. Together they were to provide water to irrigate 405,000 ha (1.0×10^6 acre) authorized the construction of facilities to provide for drainage. The voter-approved State Water Project authorized State participation in a drain if drainage repayment contracts were signed by users. Public Law 86-488 authorizing construction of the federal San Luis Unit, also mandated either participation with the state in a master drain or construction of an interceptor drain to collect irrigation drainage water from the federal San Luis Unit service area and carry it to the delta for disposal. Unfortunately, no final proposal had been prepared and there was a recognized need to jointly study drainage needs with the state. This began a long process of developing a proposal for consideration.

"A second major change occurred during the same time period with the increased recognition that water pollution nationwide was limiting the beneficial use and development potential of many streams and rivers. Protection of the quality of existing water supplies was recognized as critical to the success of these and other projects that attempted to increase the intensity of water resources utilization. For the federal CVP, questions were being raised about the potential effects of the discharge of untreated agricultural drainage on the water quality in the Delta and San Francisco Bay (SJVDP 1990).

"This concern was reflected in a rider to the CVP appropriation act by the U. S. Congress in 1965, which stated that:

> [. . .] the final point of discharge for the interceptor drain for the San Luis Unit shall not be determined until development by the Secretary of Interior and the State of California of a plan which shall conform with the water quality standards of the State of California as approved by the Administrator of the Environmental Protection Agency (Johnston et al. 2011).

"A similar rider has been included in every annual CVP appropriation act since 1965 (SJVDP 1990)."

Agriculture in the San Joaquin Valley had gained international acclaim for its diversity and productivity. Cultivation practices and irrigation technology evolved continuously, as farmers responded to changes in social perceptions and public priorities regarding natural resources, the environment, and market constraints. Yet by the end of Phase 1, several environmental issues arose that threatened irrigation sustainability using the existing production methods.

2.2.2 Phase II. 1967–1982 (Table 2.2)

2.2.2.1 Continued Expansion Amid Initial Environmental Concerns

During this period, 1967–1982, farmers and public officials in California moved slowly to address salinity and drainage issues. New laws at the state and federal levels enacted to protect water quality and the environment impacted irrigation in California. In 1967 (Table 2.2), nitrates in drainage water were identified as a concern in the northern portions of the agricultural drainage area and the Sacramento-San Joaquin delta region (Letey et al. 1986; Johnston et al. 2011). Work done between 1982 and 1984 near Los Banos demonstrated that practical methods existed that could be used to remove nitrate in the agricultural drainage. In 1968, the State of California withdrew its support for a master drain. But the nitrate problem had delayed the completion of a canal to serve the CVP, the San Luis Drain (Letey et al. 1986). Another very noteworthy aspect of the nitrate problem was that it initiated a change in the perception of irrigated agriculture: an awareness among many that disposal of agricultural drainage water could cause significant environmental problems.

Porter-Cologne Water Quality Control Act (1969)

In 1969, the state legislature passed the Porter-Cologne Water Quality Control Act to preserve, enhance, and restore the quality of the state's water resources. Before Porter-Cologne Act was passed, California did not have a statewide regulation on the discharge of agricultural and industrial wastewater. Porter-Cologne Act established the State Water Resources Control Board (SWRCB) and gave it full authority over water quality in the state's rivers, streams, and reservoirs; mandated the establishment of nine regional water quality control boards throughout the state; and required the regional boards to prepare water quality plans and objectives to ensure beneficial use of the state's water supply and to ensure environmentally safe discharge of spent water. Water quality objectives included setting specific limits on the allowable concentrations of environmentally harmful constituents in agricultural wastewater. The discharges of pollutants from point and non-point sources, such as agriculture, were regulated.

California Environmental Quality Act (1970)

California Environmental Quality Act (CEQA) stipulates that state and local agencies within California must identify and publically disclose significant environmental impacts of their actions on proposed projects and adopt feasible measures to mitigate those impacts. Permits to appropriate water in California have been subject to compliance with this statute since the enactment. Emergencies represent an exception when water transfers are exempt from CEQA review.

San Luis Drain, First Stage (1971)

Pursuant to the provisions for drainage in the federal San Luis Authorizing Act of 1960, the U.S. Bureau of Reclamation proceeded with the first stage of construction of the San Luis Drain, which was completed in 1971, originating on the eastern boundary of the Westlands Water District (WWD) and terminating next to a series of shallow, excavated flow-regulating ponds that were subsequently named Kesterson Reservoir.

Federal Clean Water Act (1972)

Three years after California passed Porter-Cologne Act, the federal government enacted the Water Pollution Control Act of 1972, also known as the Clean Water Act to maintain water quality nationwide and to prevent excess discharge of pollutants into the nation’s water supply. The federal government can enforce federal water quality standards in California via the Clean Water Act, which set ambient water quality standards, limiting the concentrations of pollutants within the nation’s surface water bodies.

Pollutant Discharge Permits (1972)

The Clean Water Act set effluent standards and established a permit system, regulating the quantity of pollutants discharged from a “point source,” such as a pipe, ditch, or tunnel, into navigable bodies of water. Although federally mandated, permits associated with the National Pollutant Discharge Elimination System (NPDES) are administered in California by the State Water Resources Control Board (SWRCB). “Non-point source” water pollution is subject to federal regulations and California state laws that are also administered by the SWRCB. The discharge limits are defined as total maximum daily loads (TMDL) for each pollutant named. Discharges of pollutants in the aggregate from multiple agricultural and irrigation operations, as in the San Joaquin Valley, are described as “non-point source” pollution because one, specific source cannot be identified or singled out from the aggregate. Agriculture is the single most important factor in non-point source pollution, gradually degrading the terrestrial environment near its production acreage and impacting downstream water quality.

Installation of Subsurface Drains

As the irrigated area in the San Joaquin Valley expanded in response to new water supplies (Fig. 2.1), the potential problems of salinity and drainage increased. The virgin soils were calcareous, and reflected a wide range of salinity, sodicity, gypsum, and boron levels because of their marine origin. The salinity and drainage problems predicted by Hilgard (1886) became a constraint on agriculture. Installing subsurface drains provided the means to control both water logging and soil salinity. Reclamation was achieved by applying more water than needed by the first crops grown, which usually were field crops, such as barley, wheat, sorghum, cotton and sugar beets that tolerate both salinity and boron (Dean's Committee, Univ. California 1968). As long as the water table remained 1–2 m below the root zone, salts moving downward through the soil to underlying soil strata as a result of reclamation did not pose a problem. When reclamation and subsequent farming result in shallow water tables, the normal practice is to install subsurface drains at depths below the water table to intercept the water percolating downward below the root zone of the crop and to transport the intercepted water – which contains nutrients, salts and chemical residuals – away from the field.

Since subsurface drains are installed only in areas of water districts where the water table is very close to the land surface, the tile drained networks occupy only limited parts of each water district. For example, in the WWD, the initial subsurface drainage system was installed just south and west of the city of Mendota on about 17,000 ha (42,000 acres) or slightly more than 6 % of the irrigated lands within the district. This system consisted of field drainage laterals, sumps into which these laterals discharged, and collector drains, which conveyed the saline drainage water from the farm sumps into the San Luis Drain.

On-farm drains in the irrigation and water districts north of Mendota were installed as early as 1951 when the Delta Mendota Canal brought imported water to those areas. The drainage waters from these drains were historically conveyed to the San Joaquin River through a network of shared open drains, canals and ditches. However, water districts south of Mendota, such as WWD in Kings County and those in Kern County all located within the Tulare Lake Basin, had neither a natural discharge point nor a legal permit that allowed discharge to the River. (Discharge permits did not exist when these drains were installed because these drains predated water quality control regulations or basin plans that addressed agricultural drainage issues, as required by Porter-Cologne Act and the Federal Clean Water Act. in 1969 and 1972, respectively.) Construction of subsurface drains in these water districts required a master drain to carry saline water out of the hydrologically closed Tulare Lake Basin. In 1975, federal and state governments formally initiated plans for a master drain to collect and transport subsurface agricultural drainage from the west side to Suisun Bay in the Delta (the San Luis Drain) for areas served by the CVP and the SWP. The initial segment of the proposed master drain had been constructed. See San Luis Drain (1971).

National Wildlife Refuge Administration Act (1976)

In 1976, with the passage of the amended National Wildlife Refuge System Administration Act, all wetlands managed by the U. S. Fish and Wildlife Service were incorporated into a national wetlands system. Kesterson Reservoir and the adjacent 1,870 ha (4,620 acres) were subsequently incorporated into the Kesterson National Wildlife Refuge, which served as a federally protected habitat for domestic shore birds and migratory birds of the Pacific flyway. Between 1972 and 1978, Kesterson Reservoir received fresh water inflows. Due to financial and regulatory reasons, construction of the second-phase, the lower reach of the San Luis Drain, from the Kesterson Reservoir to Guistine, was abandoned in 1979. Discharges of agricultural drainage water from about 2,150 ha (5,300 acres) of the WWD commenced in 1980, and in 1982, all inflows into the Kesterson Reservoir consisted exclusively of agricultural subsurface drainage water, which is critical to understanding the subsequent environmental problems related to selenium that began the following year (1983) during Phase III of this chronology.

Sustainability Assessment (1982)

Without the benefit of hindsight, by 1982, stakeholders believed the local drainage problem in the San Joaquin Valley was being managed and was on the road to being resolved. The availability of low-cost, low-salinity water supplies for irrigation, combined with the installation of subsurface drains to control water logging and soil salinity, and coupled with the construction of the first stage of the San Luis Drain, together appeared to ensure the sustainability of irrigated agriculture along the west side of the San Joaquin Valley. Irrigation management was also improving, reducing water application rates. Technology was making it possible to continue federal and state policies that supported intensive, high-value irrigated agriculture. Under the assumption that the second phase of the San Luis Drain would be completed at a later date, even though it had been abandoned initially in 1979 (see above), the San Luis Drain/Kesterson Reservoir drainage system was assumed to have an environmental benefit: it was a suitable wetland area for waterfowls and shorebirds. But the second phase of the drain was not completed in a timely manner, evaporation was not as rapid as anticipated, and the unexpected occurred. . ..

2.2.3 Phase III. 1983–1990 (Table 2.3)

2.2.3.1 Selenium Contamination at Kesterson Brings National Attention and Creates Regional Urgency

Ecotoxicosis at Kesterson Reservoir (1983)

In 1983, substantial reproductive failure of waterfowl and deformities in waterfowl embryos at Kesterson Reservoir in the San Joaquin Valley were observed and rapidly determined to have been caused by selenium poisoning (Table 2.3). During irrigation, more water is applied than the crop needs, and the extra water leaches salts downward through the soil profile to maintain soil salinity in the crop root zone at levels that do not reduce crop yields. On the west side of the San Joaquin Valley, this extra water was also leaching naturally occurring selenium from the soils derived from parent material of marine-origin. Although initial concentrations in the drainage water were not toxic, selenium was taken up by lower trophic level plants and animals, and bio-accumulated along the ecological echelon to toxic levels. In 1983, many birds at the national refuge in the San Joaquin Valley (embryos, and chicks) died as a result. The selenium problem at Kesterson National Wildlife Refuge received significant media attention and called into question the safety of agricultural drainage water from the western San Joaquin Valley. Several years later, the National Research Council issued a broad warning about the consequences of irrigated agriculture that is quoted in the Prologue of this book (National Research Council 1989).

Shift in Public Perception of Agriculture

The resulting news coverage characterized the selenium contamination as an environmental disaster, which served a direct blow to the urban public's perception of agriculture, shifting it from an appreciation of the food and fiber produced as a result of irrigation to a serious concern about the adverse environmental impacts of irrigated agriculture. The U.S. Fish and Wildlife Service involvement in the operation of Kesterson Reservoir, which had been incorporated into the National Wildlife Refuge System, was beneficial in that it allowed for early detection of selenium toxicosis. Lacking that early scientific contribution, the long-term chronic consequences of selenium contamination might have caused more instances of nesting failure and wildfowl mortality. The Kesterson experience helped, albeit dramatically, to focus attention on the rare environmental consequences of intensive irrigation and subsurface drainage in an arid environment. After the Kesterson crisis, other sites in the western U.S. were identified where concentrations of selenium and other trace elements, such as arsenic, occur at potentially toxic levels (National Research Council 1989).

Crisis Response by Federal Regulators (1985–1986)

Initially, the selenium toxicity to birds at Kesterson triggered a crisis response and an immediate regulatory and scientific review of existing irrigation practices, their potential impacts on water quality, and management options for addressing impacts. This evaluation was somewhat constrained by the need to rapidly acknowledge, ameliorate, and remedy the selenium toxicity issue at Kesterson. The set of management alternatives was constrained by the perceived need to act quickly and decisively.

In 1985, because wildlife were dying at the Kesterson Reservoir due to selenium toxicosis, the Secretary of the U.S. Department of the Interior (DOI), who oversees the United States Bureau of Reclamation (USBR), ordered the closure of drainage water collection ponds at the Kesterson National Wildlife Refuge. That decision, and the subsequent plugging of agricultural drains on a small area the west-side, completed in May 1986 (Tables 2.1, 2.2, 2.3, and 2.4), marked the beginning of the inevitable decline in irrigated agriculture that Hilgard predicted in his writings in 1886. Hilgard wrote that if irrigation was expanded in California without management strategies that recognized the need to use water “sparingly”, if “the necessity of drainage correlative with irrigation” was ignored, and if “whole communities” did not take timely action on these issues then “what has happened in India will assuredly happen” in the “great valley of California” (Hilgard 1886).

The DOI also decided to terminate irrigation water delivery to the 16,800 ha (41,514 acres) served by the WWD drainage collector system and to end the delivery of agricultural drainage water to Kesterson in 1985 (Table 2.3). By the end of 1988, discharge of drainage water into the San Luis Drain was stopped. The State Water Resources Control Board issued an Order (SWRCB 1987) requiring Kesterson Reservoir to be capped with a 15 cm thick layer of clean soil to eliminate aquatic food chain exposure to selenium.

University of California Salinity Drainage Task Force (1985)

The University of California (UC) responded to the Kesterson crisis in 1985 with the establishment of the UC Salinity Drainage Task Force (hereafter Task Force), a group of research scientists appointed to study the problems at Kesterson and to coordinate with water and drainage district managers and local farmers to enhance knowledge that might contribute to developing a sustainable future for agriculture in the San Joaquin Valley. One primary goal of the Task Force was to examine in-Valley solutions to the salinity, selenium, and drainage problems. In defining its research scope, the Task Force initially described the challenge:

The historic expansion of California agriculture had occurred without adequate consideration of the inevitable problems associated with salinity. Groundwater pumping in the early 1900s and subsequent surface water development in mid-century had outpaced limited efforts to address salinity and drainage. Farm-level drainage was not developed in the context of an overall, regional solution, such as a master drain to the ocean. Farmers installed drainage systems as needed to sustain productivity, without awareness of potential long-term, off-farm impacts of saline drainage water. Institutional efforts to construct a regional drainage solution were slow and inconclusive. Lacking a regional discharge option, the accumulation of saline drainage water would inevitably create a problem generating regional attention and requiring a regional solution.

Research projects funded by the Task Force involved field experiments related to irrigation management, economics, and development of local and regional scale models dealing with soil chemistry and physics, and hydrology. The Task Force examined the biogeochemistry of selenium in agricultural drainage water at multiple scales.

Shift in Public Policy Regarding Irrigation (1983–1988)

Within 5 years, a notable shift occurred in public policy regarding irrigation. Rather than seeking an engineering solution that would allow irrigation to continue thriving on the west side of the Valley by ensuring long-term drainage service, the public demanded the negative environmental impacts of irrigation to be minimized. From this time forward, irrigated agriculture in California would need to achieve compliance with rigid environmental guidelines.

Consequences of Inadequate Drainage Facilities: Analysis (1988)

The failure to provide adequate drainage to serve both the CVP and the SWP service areas has been characterized as a tragic planning error. Ron Robie, the former Director of the California Department of Water Resources, summarized the problems in obtaining approval for a Master Drain:

> In the Kern County area we [the State of California] did it wrong. The Bureau [of Reclamation] did it right, we did it wrong. We sold the water and created the [State Water] project without providing for the Drain. We had it in the law but we didn't fund it. At least the Bureau of Reclamation and Westlands Water District built the Drain into the system. Now, they left part of it off by not providing an adequate area of disposal, but at least they put the drainage in with the water at the same time, which was right. We didn't. We built the State Water Project and we had a drain on paper but nowhere else. The problem is that the drainage all goes to the center of the Valley, and various people contribute to the drainage in different amounts. If you only build a drain by charging people whose water actually goes into the drain, then you're letting a lot of people off and it's unfair. So what you really needed was some kind of taxing entity or some kind of method of determining a zone of benefit to pay for a drain. So that was the first problem, how you finance the Drain, assuming you could build one. The second problem, of course, was the .... attitude in Contra Costa County that no drain could ever go there. We did make some studies that showed that in truth and fact if you discharged the drain water way down [the Delta] in Contra Costa County, the impact on the bay [San Francisco Bay] would be relatively modest. (Robie 1988)

These and similar reflective remarks reveal an underlying discord among agricultural, urban, and environmental objectives that was intensified by the selenium crisis. The onus of solving this problem was placed on the agricultural community. Kelley and Nye (1984) summarize conflicting responses of some interests to the proposal for the San Luis Drain:

> Like its predecessors, this plan [for the San Luis Drain] was condemned [considered unacceptable] by [San Francisco] Bay Area interests and rejected by state service area—SWP – irrigators as too costly. Many potential Valley Drain users south of Kettleman City believed they could solve their immediate drainage needs for the next 20 to 30 years with evaporation ponds and other local salt disposal methods already in use. Efforts to overcome salinity and drainage problems in California have, in general, been highly successful. As these problems have advanced beyond the individual and local levels, however, and affected the interests of many divergent economic and political groups, finding adequate solutions has become increasingly complex and expensive.

In addition to the conflicts among urban, environmental, and agricultural interests, the drainage crisis was not equally distributed within the San Joaquin Valley agricultural community. While selenium is found naturally in soils on the west side of the Valley, soils on the east side generally are selenium deficient. Yet the east side had its share of drainage issues from 1880 to 1920 as described in Phase I of the irrigation chronology. However, the conjunctive use of surface water and groundwater for irrigation controls water logging and soil salinity. This practice on the east side is sustainable without the need to dispose of saline drainage water within or outside the San Joaquin Valley because the surface water supply, which originates in the Sierra Nevada, is naturally low in salinity (<0.1 dS m^{-1}), and the native soils derived of the Sierra Nevada do not contain selenium. Thus, the drainage and selenium crisis did not affect east-side farmers.

San Joaquin Valley Drainage Program Report (1990)

The San Joaquin Valley Drainage Program (SJVDP), which included technical staff from state and federal agencies, was established to begin studying the implications for public policy regarding drainage service in the aftermath of the initial responses to the Kesterson problem. A comprehensive list of solutions was developed to reduce drainage water volume on farms and to remove selenium from drainage water. None of the proposed solutions would be cost-effective in reducing selenium concentrations and loads sufficiently to achieve state and federal water quality guidelines. In its final report, the SJVDP described a range of possible actions that could be implemented to reduce selenium concentrations and loads:

- Source control;
- Drainage water reuse;
- Evaporation; protection, restoration, and provision of substitute water supplies for fish and wildlife habitat; and
- Institutional change.

2.2.4 Phase IV: 1991–Present (Table 2.4)

2.2.4.1 Search for Sustainable Solutions to Salinity, Selenium, and Drainage

Science-Based Solutions (1991)

The Kesterson crisis triggered an on-going program of research and management of drainage from irrigated agriculture in a closed basin. With funding from the California Legislature and local agencies, the San Joaquin Valley Drainage Implementation Program (SJVDIP), established in 1991, brought University of California researchers, water and drainage district managers, and local farmers together in an effort to better understand the drainage issues and management options for addressing them within the Valley. The UC Salinity Drainage Task Force provided a core scientific team to guide this cooperative effort. Looking at the drainage problem in the context of the selenium crisis at Kesterson, this Task Force examined the biogeochemistry of trace elements in agricultural drainage water at multiple scales: laboratory, individual plant (greenhouse), field, and watershed or water district scales. The basic research led to field experiments in irrigation and drainage management to test and refine the findings of the review mentioned in the previous section.

Cooperation among researchers funded by the Task Force, and the staffs of DWR and SWRCB was fostered by annual meetings held for the purpose of reporting results from funded projects, and for providing a forum where proponents of methods to treat selenium, desalination, and evaporation pond management with mitigation presented their views. Research scientists of the Task Force, together with staff members from DWR and SWRCB participated in the activities to update the management plan proposed by the interagency program comprised of federal and state agencies (SJVDP 1990). Eight UC faculty expertise groups were tasked with technical and economic evaluation of the management options proposed. The topics addressed were drainage reuse, drainage treatment, land retirement, evaporation ponds, source reduction, groundwater management, river discharge, potential uses of drainage waterborne salts and selenium.

Integrated On-Farm Drainage Management

Following the Kesterson crisis, the prospect of discharging agricultural drainage water outside the San Joaquin Valley diminished substantially. Considerable opposition had formed in the urban and agricultural sectors regarding the potential impacts of selenium-tainted drainage water in the San Joaquin River, its tributaries, and the San Francisco Bay. Hence, the goal of the Task Force, in coordination with the SJVDIP, was to examine in-Valley solutions based on an integrated, on-farm drainage management program (IFDM).

The salt management option available to individual farmers, known as IFDM, sets the goal of eliminating discharge of saline drainage water from a farm to surface water bodies. Where suitable subsurface drainage water (i.e. total dissolved solids of water, TDS $< 6{,}000$ ppm or electrical conductivity of water, EC < 8dS m^{-1}) is available, it can be re-used for irrigation of salt-tolerant crops, which increases irrigation efficiency and reduces the volume of drainage water needing disposal using the IFDM technology (Cervinka et al. 1999; Wichelns 2005). The key piece of equipment of an IFDM program is a solar evaporator, used to evaporate a small volume of highly concentrated saline drainage water, leaving behind a very thin layer of salt to be collected and disposed. The evaporator resembles a condensed version of the larger scale evaporation pond. To extend the useful life of the solar evaporator as long as possible, the volume of drainage water requiring evaporation must be minimized. This is accomplished primarily by using drainage water to irrigate salt-tolerant crops and halophytes, in succession, before collecting and evaporating the remaining drainage water from the last stage of this crop irrigation sequence. Advantages of IFDM include the opportunity it provides individual farmers to manage salts and irrigation water on their farms. Potential disadvantages include the need to set aside potentially productive land on each farm for the evaporator and for the salt-tolerant crops and halophytes, and the farm-level costs of building and operating the solar evaporator. The operating cost of IFDM likely will decline over time as farmers in a region gain experience with the program and learn how to minimize the amount of land required for non-marketable crops. Given the high incremental value of water in the San Joaquin Valley, use of reverse osmosis to recover useable water from the concentrated drainage water generated by IFDM is currently under active testing. Selenium would be removed from the brine water generated by reverse osmosis before the water would be delivered to solar evaporators or evaporation ponds.

On each individual farm, adoption of the IFDM program would accomplish the following five objectives:

- Reduce drainage water volume requiring treatment or disposal at the source
- Further improve irrigation methods
- Increase the re-use of drainage water
- Retire additional land
- Dispose of the remaining drainage water in evaporation ponds or deep groundwater basins

These five objectives of IFDM are interwoven and dependent, as will become clear in the detailed discussion that follows. Progress on one front depends on and enables progress on another front, until the volume of subsurface drainage water requiring treatment and/or disposal poses a greatly reduced threat to the environment.

Reduce Drainage Water Volume Requiring Treatment or Disposal at the Source

Substantial research has been conducted regarding source reduction through improvements in irrigation methods, such as better management of surface delivery systems and conversion to micro-irrigation, and drainage water reuse. In some areas, when the cost of successfully improving irrigation management or increasing the re-use of drainage water exceeds the incremental gain, it becomes optimal to cease irrigation and retire the land, or use it only for dry land rain-fed crop production. In some cases, the declining availability of irrigation water for agriculture also contributes to the decision by landowners to discontinue irrigated farming.

Further Improve Irrigation

Beginning in the 1970s, researchers sought ways to reduce drainage water volume in both small (van Schilfgaarde et al. 1974; Rhoades et al. 1974; Oster and Rhoades 1975) and field-scale projects (Hoffman et al. 1984; Rhoades et al. 1989). Their findings supported the concepts of increased application efficiency, minimum leaching, and salt storage within irrigated soils. Drip irrigation and linear-move sprinkler systems are most effective in the areas they studied, although the net economic impacts vary among crops. The farm-level economic prospects of adopting higher technology systems will improve as farmers continue switching from lower valued field crops to higher valued fruits and vegetables, with changes in market conditions and increasing water scarcity. Hanson et al. (2006) obtained profitable yields producing tomatoes with drip irrigation on saline soils in the San Joaquin Valley.

Increase the Re-use of Drainage Water

Drainage water reuse can be increased by (a) blending drainage water with canal water deliveries, (b) sequentially irrigating salt-tolerant crops with drainage water, and (c) designing subsurface drainage systems to enable active management of shallow water tables. In some areas, drainage systems can be modified to enhance crop use of shallow groundwater during the crop season, while providing drainage relief during other times of the year. Drainage water volumes and salt loads can be reduced substantially by actively managing subsurface drainage systems. Sequential re-use of drainage water on crops of increasing salt tolerance can be enhanced to remove selenium from the drainage water by allowing selected organisms to accumulate selenium. The method is known as serial biological concentration (Rhoades et al. 1989; Cervinka et al. 1999; Kaffka et al. 2002).

Retire Additional Land

Large areas of crop production land have already been retired in the San Joaquin Valley, due partly to reductions in surface water availability and partly to the difficulty of maintaining productivity without adequate drainage service. In 2005, all landowners in the 3,845 ha (9,500 acres) Broadview Water District agreed to sell their land to the WWD, in a transaction that enabled WWD to obtain Broadview's contractual water supply (Sidebar 2.3). The persistent shortage of water together with regulatory action against WWD has contributed to the retirement of about 40,469 ha (100,000 acres) of land in recent years. Much of the retired land is the "drainage problem area" within WWD. It is likely that additional land will be retired in WWD and other districts in the northern portion of the San Joaquin Valley, as farmers and districts continue their efforts to achieve zero-discharge of subsurface drainage water to surface waters.

Sidebar 2.3 Broadview Water District: Drainage Complications

The history of the Broadview Water District, on the west side of the San Joaquin Valley, exemplifies irrigation development in the 1950s followed by decline in the early 2000s, due largely to environmental problems posed by disposal of drainage water. This District was formed in the 1950s by farmers wishing to obtain a surface water supply for irrigating 4,000 ha (10,000 acres) of farmland in central California. The farmers had been using groundwater from a few deep wells in the area, but the salinity and boron concentrations in the groundwater were higher than desirable for long-term irrigation. When the District obtained a contract from the CVP for delivery of water from the Delta-Mendota Canal, the wells were abandoned and farmers began irrigating with surface water.

The District's annual contract supply of 33 million m^3 (27,000 acre-feet) from the CVP was sufficient to irrigate one grain crop in winter and to produce one primary crop in summer. Winter grain crops included wheat and barley, while primary crops included cotton, tomatoes, cantaloupes, seed alfalfa, and sugar beets. To obtain the needed drainage to control water table levels and soil salinities, on-farm subsurface drainage systems were installed in about two-thirds of the district between 1960 and 1980. Lacking a discharge outlet for disposal of the drainage water, the District recycled all of its commingled surface runoff and subsurface drainage water by combining it with fresh water deliveries. Over time, the salinity of water deliveries increased, as did the salinity levels in farm fields (Wichelns 2003).

Upon obtaining a discharge outlet in 1983, the District began releasing its commingled drainage water into open drains that flowed toward the San Joaquin River. Some of this saline drainage water was used by farmers and duck club owners to supplement their irrigation water supplies, while some of it likely reached the San Joaquin River. Following the Kesterson event, the State Water Resources Control Board (1987) began regulating the volume of drainage water and specifically the amount of selenium in the water that could be discharged by Broadview and other districts in the region. The permissible amount of selenium that could be discharged was scheduled to decline, over time, to levels that would require Broadview to again recycle most of its commingled drainage water. Having already had the negative experience of such a recycling regimen, all of the landowners in Broadview decided in 2005 to sell their land for the value of its water allocation to Westlands, a neighboring water district (Wichelns and Cone 2006).

Dispose of the Remaining Drainage Water and Its Associated Salts in Evaporation Ponds, Designated Receiving Lands, or Deep Groundwater Basins

Given the goal of zero salt and selenium discharge to surface water bodies, farmers and water districts must develop alternative disposal strategies for subsurface drainage to sustain productivity. Farmers in the Tulare Lake Drainage District have been discharging drainage water to large, regional evaporation ponds for many years. The California Regional Water Quality Control Board (CRWQCB) ensures that the District complies with environmental regulations pertaining to three issues: (a) the pond size and the lining required in prevent groundwater pollution; (b) the volume of drainage water stored in the ponds; and (c) the concentration of selenium in the ponds. The Board requires that Se concentrations of water in the evaporation ponds remain below 5 μg L^{-1}, or solar evaporator technology is required, which in turn, requires that potential evaporation rates exceed actual ambient evaporation rates; i.e., evaporation takes place more quickly than the rate at which drainage water is generated. For evaporation ponds to continue operating, Se levels must remain below 5 μg L^{-1}, which means Se must be removed before the water is discharged into the evaporation pond. Regulatory agencies may require pond operators to construct managed wetlands near evaporation ponds to mitigate the pond's harmful effects on wildlife.

2.2.5 IFDM Programs Scaled Up to Regional Scale in the San Joaquin Valley

To make the IFDM effective for the San Joaquin Valley region, the Task Force would need to address five about referenced objectives at the larger perspective of an integrated, regional, in-Valley solution and at varying spatial and temporal scales. Consistent with the scientific development, farmers and water managers would need to approach drainage management from the perspective of both individual farm scale and regional scales. Therefore, larger-scale regional programs that support IFDM include the following features:

2.2.5.1 Discharge Drainage Water to Dedicated Receiving Lands

Several farmers and water districts within large-scale irrigation schemes discharge saline drainage water to lands that have been dedicated for that purpose (Quinn et al. 1998; Murry-Darling Basin Ministerial Council 2001). Farmers in the Grassland area of the San Joaquin Valley have been discharging drainage water to a 1,538 ha (3,800 acres) parcel of land near the Panoche Drainage District for many years. The group discharged 13.6 million cubic meters (11,000 acre-foot) of drainage water there in 2007, using the water to produce salt-tolerant crops, including Jose Tall Wheatgrass (McGahan and Falaschi 2008). The Grassland area farmers plan to increase the size of their dedicated receiving lands to 2,428 ha (6,000 acres) as part of their program to achieve zero-discharge to surface waters.

2.2.5.2 Discharge to Groundwater

At a regional scale, achieving zero-discharge will require that excess salts, which would have been discharged to the San Joaquin River, accumulate and become stored in soils and groundwater. At first glance this approach might seem non-sustainable. Yet, if aggressive source reduction efforts are undertaken and groundwater is managed wisely, and if a usable water quality target of TDS = 2,500 ppm TDS is adopted, then groundwater in the San Joaquin Valley might provide a viable storage site for salts (Belitz and Phillips 1995). The usable water target could be increased to TDS = 6,000 ppm in some applications (Letey et al. 2003; Oster 1994). The use of this approach might be limited in areas where groundwater pumping creates a downward hydraulic gradient causing downward movement of drainage water generated by irrigation. Groundwater pumping lowers the water table, reducing the land area needing subsurface drainage systems. Fogg et al. (1999) noted:

> The concept of mitigating the drainage problem by groundwater management is, in principle, very simple. The high water table results from an imbalance in the water budget – water is being applied to the surface at a rate that exceeds the carrying capacity of the groundwater system, thereby raising groundwater levels. In groundwater management, groundwater pumping is increased in order to remove the excess groundwater and lower the water table. The water budget can be further modified by reducing groundwater recharge via decreases in applied water.

A universal conclusion of farmers, water managers, and Task Force scientists is that some combination of these methods for source control and management of residual drainage water will be essential in developing a zero-drainage solution.

2.3 Policies and Institutions Need to Support Technology Adoption

To implement an in-Valley salt management program at farm and regional scales, supporting policies and institutional framework are needed to complement the models, technology, and resources required. Farmers need incentives to internalize the external impacts of their irrigation and drainage activities. Salinity permits or charges could be designed to motivate farmers and district representatives to modify their drainage water management practices in ways that reduce or eliminate discharge to surface waters. Policies could be implemented in conjunction with surface water delivery rights and regulations regarding groundwater withdrawals could be strengthened.

Substantial political effort might be required to implement innovative water and salt management policies in the San Joaquin Valley, particularly given that water rights and resource endowments vary widely among farmers throughout the region. However, the goal of achieving sustainable irrigation at reasonable cost is shared by all farmers. The likelihood of achieving that goal probably is maximized when farmers form regional associations that can implement effective drainage management programs. Those efforts will involve features of multiple IFDM programs, farm-level improvements in irrigation practices, re-use of drainage water, regional collection and disposal of drainage water, and regional management of shallow and deep groundwater.

Irrigation and drainage districts in the Grassland area have begun implementing a long-term program they call the Westside Regional Drainage Plan (Quinn et al. 1998). Westlands Water District and the U.S. Bureau of Reclamation have begun discussions of a plan in which WWD would assume responsibility for developing drainage systems and treatment facilities to dispose treated drainage water within the District. And, beginning with the land retirement in the early 1990s, WWD continues to purchase and 'retire' irrigated lands in areas with high water tables. The premise of these efforts is the same: farmers and their district personnel must develop an in-Valley strategy for managing salt, while complying with environmental standards, given that ocean disposal via the Delta is not a viable policy option. Both efforts will involve further improvements in irrigation practices, more aggressive salt management, and retirement of a substantial amount of farmland in selected areas. The cost of implementing these strategies will be quite large, but necessary, to sustain irrigation in the Valley. A long-term, sustainable management program is needed to ensure that agricultural productivity can be sustained on a subset of the Valley's currently irrigated lands, while not causing undue harm to the region's land, water, and wildlife.

2.4 The Search Continues for a Sustainable Solution

Irrigation and drainage development have moved through four stages in the San Joaquin Valley:

- Long-term development of groundwater pumping, in advance of surface water deliveries,
- Rapid expansion of irrigated area with the availability of imported surface water, although with limited drainage service,
- Moderate expansion of farm-level drainage systems, with inadequate development of regional drainage infrastructure, and
- The search for in-Valley salt management options, given that out-of-Valley salt disposal will not be possible.

Several observers, including E.W. Hilgard and J. van Schilfgaarde (Skaggs and Van Schilfgaarde 1999), suggested long ago that irrigation can be sustained only if proper management of water and salts is implemented at the outset and maintained. Ideally, farmers, water agency managers, and public officials should have heeded the warnings and implemented a sustainable drainage system at the same time that irrigation schemes were developed in the San Joaquin Valley. It is too late to modify the history that has unfolded as Professor Hilgard predicted. Water tables have risen, lands have become saline, and the volume of saline drainage water requiring disposal or re-use each year is now substantial. Selenium has greatly complicated the current situation and reduces the number of feasible alternatives to the Valley's drainage problem. Irrigation can be sustained only if proper management of water and salts is implemented.

The cost of sustaining irrigation, as results of both water and drainage management costs, may ultimately be too high, particularly in areas with stringent environmental standards involving protections of land and water quality (Wichelns and Oster 2006). Even with improved scientific understanding of the potential in-basin solutions to salinity and drainage, both political and economic forces may at the end determine the success of irrigated agriculture in the San Joaquin Valley.

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Chapter 3
The San Joaquin Valley: Salinity and Drainage Problems and the Framework for a Response

Nigel W.T. Quinn

The problems of irrigated agriculture in arid, drainage-impaired basins around the world share many characteristics. As a case history, Westside of the San Joaquin Valley provides insights into how these age-old problems are addressed in a context of changing values and priorities and an increasingly complex regulatory environment, complicated by an unexpected selenium eco-toxicity issue. The lessons learned in addressing the issues have applicability to similar drainage-impaired situations worldwide. For example, the models developed to allow growers and managers to optimize their irrigation and drainage systems may be adapted and engineered for systems with similar characteristics. Regulatory responses to the salinity- and selenium-induced environmental problems in California are likely similar to those that would be enacted if similar circumstances were to occur in other regions of the United States. However, the experience of the University of California (UC) Salinity/Drainage Task Force is uniquely Californian and provides some insight into the physical, economic, and institutional problems in resource management in California. For general readers and readers with a practical interest in San Joaquin Valley issues, it is important to understand the physical, economic, and policy setting in which the research work of the Task Force occurred.

3.1 San Joaquin Valley Geology

The San Joaquin River Basin extends roughly NNW-SSE, descending from the foot of the Tehachapi Mountains, northwards to its confluence with the Sacramento River in the Sacramento-San Joaquin Delta. The approximately 400 km (250 mi)

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_3, © Springer Science+Business Media Dordrecht 2014

long and 80 km (50 mi) wide San Joaquin Valley is bounded on the east by the Sierra Nevada Mountains and on the west by the California Coast Ranges.

There are two major hydrologic basins in the San Joaquin Valley. On the east and north of the San Joaquin River, the San Joaquin Basin extends south of the Sacramento – San Joaquin Delta and is naturally drained by the San Joaquin River. The Tulare Basin extends from the foot of the Tehachapi Mountains northwards to a gentle topographic rise along a line a little to the north of the Los Gatos Creek and the Kings River. The Tulare Basin has no drainage outlet. A fresh water lake, the Tulare Lake, occupied this depression at the turn of the century (Preston 1979) and has since been converted to agricultural lands. The Kings River, the Kaweah River and the Tule River drain into the Tulare Lake. Further south, the Kern River empties into the smaller Buena Vista Lake. Lake Isabella is a reservoir on the Kern River, upstream of Bakersfield. The Pine Flat Reservoir is located on the Kings River. Fresno Slough connects the Tulare Basin with the San Joaquin River, across the topographic rise. Depending on the severity of flooding, when it occurs, water may flow either way in this channel.

The major tributaries to the San Joaquin River that drain the eastside of the San Joaquin Basin are the Fresno, Merced, Tuolumne and Stanislaus Rivers. The headwaters of these rivers contain high quality water in terms of dissolved salts that is important in providing dilution to the San Joaquin River when it receives poor quality subsurface drainages from Westside sources. Many surface water impoundments have been created by flood control and water supply dams on the tributaries of the San Joaquin River, including Millerton Lake (on the San Joaquin River), Hensley Lake (on the Fresno River), Lake McClure (on the Merced River), New Don Pedro Lake (on the Tuolumne River) and the New Melones Lake (on the Stanislaus River). On the Westside of the San Joaquin Basin, the major facility is the man-made San Luis Reservoir, which is connected hydraulically to the California Aqueduct and the Delta Mendota Canal and which provides off-stream storage for both the Central Valley Project (CVP) and the State Water Project (SWP).

Prior to construction of the CVP and the Delta Mendota Canal, San Joaquin River water was used to irrigate land on Westside of the San Joaquin Valley. The CVP provided for water released from Friant Dam to be diverted to the Tulare Basin through the Friant-Kern Canal; water from the Delta Mendota Canal was made available to "exchange contractors" for irrigating lands formerly served by the San Joaquin River. The Mendota Pool is a CVP storage reservoir located at the terminus of the Delta Mendota Canal, which provides water to a number of supply canals that take the water north to irrigation turnouts located along their length.

Belitz (1988, 1990) provides an account of the state of the groundwater system as it existed at the turn of the century, elucidating how its character has changed up to the present time due to intensive irrigated agriculture. Although his account addresses the central part of the Westside, largely falling within the drainage areas of the Panoche Creek and Little Panoche Creek, the ideas and issues presented can be applied reasonably to the entire Westside. Because the regional groundwater system is driven by gravity, physiography and geomorphology play a decisive role in determining its character. The Westside is characterized by a fairly simple topographic pattern: an easterly sloping flank of the Coast Ranges extending for more than 125 km (80 mi) in the NNW-SSE direction. The distance from the boundary of the Valley deposits to the San Joaquin River is about 32 km (20 mi), but varies slightly. Along this distance, the elevation declines from about 182 m (600 ft) to about 49 m (160 ft) mean sea level (MSF). The upper slope (comprising of alluvial fans), from 182 m (600 ft) to about 91 m (300 ft), tends to be steeper than the lower slopes. Four intermittent streams (from south to north, Los Gatos Creek, Cantua Creek, Panoche Creek, and Little Panoche Creek) have well-developed alluvial fans.

Sediments of recent alluvium deposited by the action of these four streams cover much of the Westside, from the flanks of the Coast Ranges to the vicinity of the river. On the upper slopes and in the prominent alluvial fans, the sediments tend to be coarse-grained, having been deposited by episodic, high-energy stream flows. In the inter-fan areas and in the lower slopes of the Valley, the sediments show a flood-plain depositional character and consist of fine-grained materials. Mass-wasting, mud flows and surge flows associated with the high energy sediment transport of ephemeral and intermittent streams appear to play a very important role in controlling the physical and chemical properties of the sediments.

Marine sediments, ranging in age from Jurassic to Miocene, are exposed along the ridge crest of the Coast Ranges (Presser et al. 1991). Two members of this sequence, the Moreno Formation (upper Cretaceous to Paleocene) and the Kreyenhagen Formation (Eocene to Oligocene) are exposed over a 32-km (20-mi) stretch of the Moreno Ridge. Despite their limited extent, these formations play an important geochemical role because of their trace elements, including selenium, boron, and arsenic. After the Miocene, during the Pliocene and Pleistocene periods, the marine conditions gave way to continental and lacustrine conditions. The Tulare formation of Pliocene to Pleistocene age underlies the alluvium over much of the Westside.

The Corcoran clay of the Tulare formation, approximately 30 m (100 ft) thick, constitutes an extensive marker horizon beneath the Westside. The alluvial sediments overlying the Corcoran clay decrease in thickness from a maximum of about 244 m (800 ft) on the Valley margins to less than 30 m (100 ft) in the vicinity of the San Joaquin River (Fig. 3.1). In the Valley trough, the coast range alluvium, characterized generally by fine-grained sediments, gives way to the alluvial Sierran sands derived from the Sierra Nevada Mountains. The coarser Sierran sands contain water with chemical characteristics distinct from the sediments of the Coast Ranges alluvium. The alluvial sediments overlying the Corcoran Clay are frequently referred to as the "semi-confined" zone.

3.2 Regional Hydrogeology

Consistent with the geology of the region, the Coast Ranges constitute the groundwater recharge area for the Westside. At the turn of the century, before intense groundwater pumping commenced in the 1920s, the piezometric heads in the deep

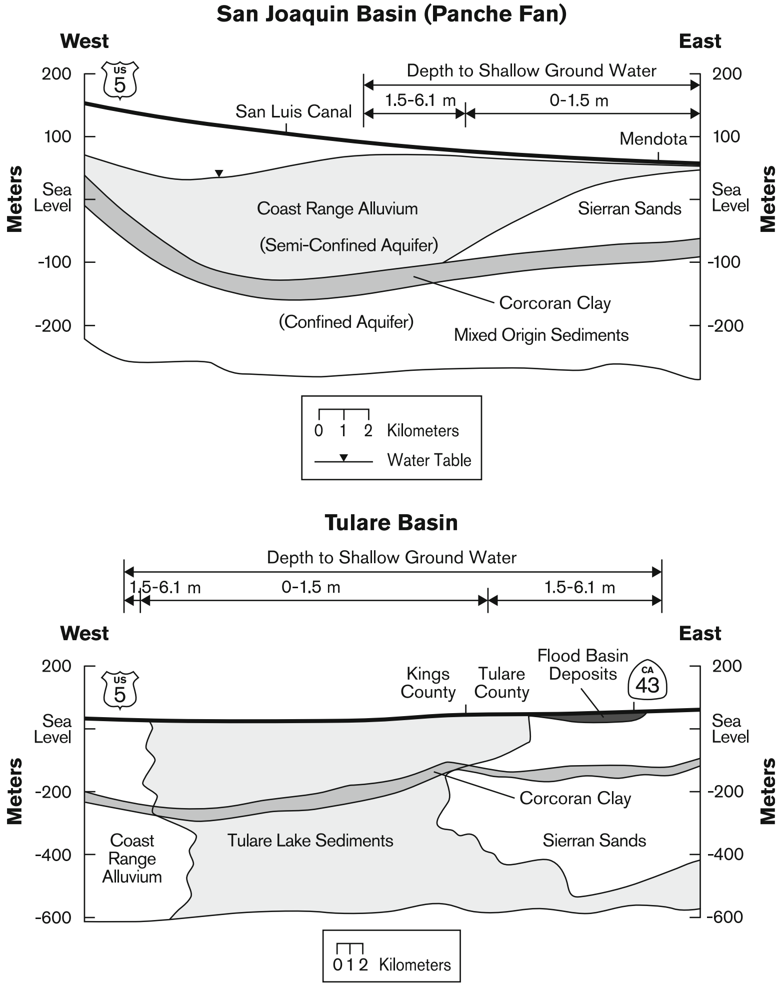

Fig. 3.1 Idealized east – west geological cross section across the western San Joaquin Valley (Belitz 1988)

aquifers underlying the Valley floor reportedly resulted in free-flowing artesian wells along a long, narrow zone of the river. The physical disposition of the artesian zone, extending parallel to the trend of the Coast Ranges, is an indication that the regional groundwater system is driven by recharge from the Coast Ranges. Based on stable isotope data of water samples from wells located above and below the Corcoran clay, Dubrovsky et al. (1990) inferred that groundwater may also be leaking vertically through the Corcoran clay and recharging the deep aquifers, both due to pervasive flow through the formation and due to the several hundred wells which are screened in horizons above and below the Clay.

The development of the west side of San Joaquin Valley for irrigated agriculture and the advent of deep-well turbine pumps in the 1920s drastically changed the groundwater flow system. Groundwater became an important component of irrigation water and, responding to post-Second-World War boom in the economy, groundwater pumping increased by a factor of four, reaching a maximum of about a million acre-feet per year (equivalent to 1.2×10^9 m^3 per year) between 1950 and 1970. Most of this pumping was from the confined aquifer below the Corcoran Clay. Water tables dropped, and there was pervasive land subsidence on the Westside that increased the cost of pumping and led to calls for a reduction in groundwater pumping. Eventually surface water from the Delta was imported via the construction of the CVP and SWP conveyance facilities.

3.2.1 Irrigation Wells

Gronberg et al. (1990) provides a summary of the distribution of wells on the Westside. Although nearly 6,000 wells are known to exist in the Valley, useable information is available only with respect to about 2,550. Nearly two-thirds of these wells are completed in the semi-confined zone overlying the Corcoran Clay. Due to the generally poor water quality within the shallow part of this zone (<16 m from land surface), most of the wells in the shallow, upper portion are passive, observation wells. Production wells in the semi-confined zone are typically greater than 16 m (50 ft) in depth. The Coast Range alluvium in the semi-confined zone generally contains fine-grained sediments. Because of the larger surface area of contact between water and solids in these sediments and longer residence times, these sediments tend to contain waters of poorer quality compared, for example, with waters of the Sierran Sands to the east of the San Joaquin River. Therefore, in the Valley trough and on the margins of the alluvial fans, irrigation wells are screened in the Sierran Sand aquifer, which contains coarser, cleaner sands and better quality water. Because of the reducing nature of these aquifers, selenium fluxes are retarded and the mobile selenate form of selenium ($SeO_4{}^{2-}$) appears to be reduced to two less mobile species, selenite or elemental selenium ($SeO_3{}^{2-}$ or Se^0). Hence, selenium concentrations from pumped wells drawing from these sands tend to be low. Wells completed in the shallow, semi-confined aquifer have screens that are typically a few meters in length.

Some 533 wells are screened in both the semi-confined zone and the confined zone, allowing communication between confined and semi-confined aquifers. Although the volume of flow between the aquifers has yet to be quantified, some hydrogeologists believe that they may account for some of the variability in the groundwater flux across the Corcoran Clay underlying the Westside. Wells penetrating the confined zone below the Corcoran Clay are generally restricted to the upslope areas at the head of the alluvial fans beyond the extent of the Sierran sands. According to Gronberg et al. (1990), 410 wells tap the confined zone with open screen intervals in excess of 30 m (100 ft). Examination by Belitz (1988, 1990)

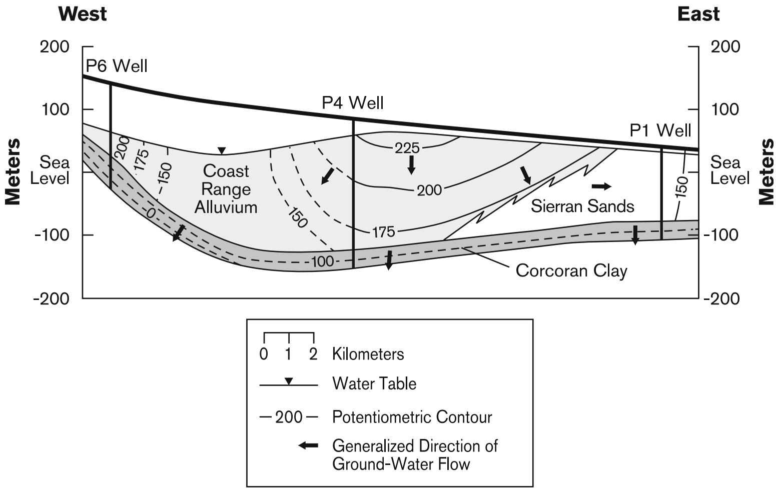

Fig. 3.2 Generalized hydrogeological cross section of the western San Joaquin Valley in 1984, showing the groundwater divide and vertical flow patterns (Belitz 1990)

of water table data and potentiometric data for 1984 showed the existence of a pronounced groundwater divide approximately midway between the Valley trough and the Coast Range. This divide, shown in cross-section in Fig. 3.2, has most likely occurred due to overdraft of groundwater by pumping to the west and by leakage from the California Aqueduct. Clearly, the pre-development areal distribution of recharge and discharge areas of the Westside have been significantly modified by the pumping and have led to a very complex groundwater hydrology and distribution of contaminants within the semi-confined aquifer.

3.2.2 Irrigation and Shallow Groundwater System

Irrigated agriculture has been practiced on the Westside for more than a century and has interacted with and modified the pre-existing groundwater flow patterns, especially recharge-area and discharge-area relationships. Application of irrigation water causes water tables to rise in the shallow semi-confined aquifer, leading to an increase in the vertically downward movement of water. Because of the large areal extent of applied irrigation water on the Westside, the resulting artificial recharge has significantly exceeded natural groundwater recharge by rainfall and stream flows. Williamson et al. (1985) estimated that from 1961 to 1977, irrigation recharge was as much as 40 times that of the natural recharge.

Applied irrigation water directly affects the shallow groundwater system. In turn, the dynamics of water flow and water table fluctuations in the shallow aquifer are intimately related to local topographic variations. Apparently, very little attention has been given by previous workers to understand these interactions on the basis of data from piezometer and tensiometer nests. Fio and Deverel (1991) studied, using nests of piezometers, the dynamic interactions between applied irrigation water, two tile drains, the water table, and the shallow aquifer at a site where drains had been in existence for 15 years. Their data, interpreted in conjunction with numerical simulations, suggest that the drains capture significant quantities of resident groundwater in the shallow semi-confined aquifer as well as deep percolation from the crop root zone immediately above the drain. Based on piezometer data down to a depth of 30 m (100 ft), they inferred that the flow direction was upward at the site from about 16 m (50 ft) below land surface and primarily downwards below 30 m (100 ft). That is, a horizontal groundwater divide seems to exist below 30 m (100 ft) depth. It is not clear, however, whether the upward flow observed by them is a manifestation of the regional flow pattern. During periods of irrigation, the gradient is reversed, causing the flow direction to change in the shallow zone, leading to downward migration of salts and soluble trace elements leached from the root zone. Grismer and Woodring (1987) investigated the importance of lateral flows to drains on the Westside from the regional groundwater flow system and concluded that the problem has to be studied on the scale of a township (intermediate scale). They also found that the data required to support accurate simulation of the regional flow system, at this scale, were not available.

3.2.3 Regional Hydrogeochemistry

The hydrogeochemistry of the San Joaquin Valley is governed by the regional groundwater flow system and by the character of the source rocks. According to Davis and Coplen (1989), the hydrogeochemistry of the Westside can be understood in terms of two distinct geochemical units. The deep aquifers below the Corcoran Clay with thickness varying from 300 to 730 m contain very old waters (615,000–725,000 years before present). These sodium sulfate waters are thought to be mixtures of waters derived from the Sierra Nevada Mountains to the east and the Coast Ranges to the west. Isolated by the poorly permeable Corcoran Clay, these waters are known to have a fairly uniform composition. The aquifers above the Corcoran Clay contain waters that are distinctly different in composition from those of the deeper aquifers. The waters of the shallow aquifers are much richer in mineral content and exhibit a great deal of spatial variability in chemical composition. The waters of the shallow aquifers can be divided into Coast Range waters and Sierran waters. In the broadest sense, groundwater in the Coast Ranges alluvium differs markedly from that in the Sierran Sands. The former contains significant quantities of nitrate, boron and selenium, while the latter is significantly higher in arsenic, molybdenum and manganese (Dubrovsky et al. 1990).

3.3 Irrigated Agriculture

Historically, agriculture on the eastside of the San Joaquin Valley has been practiced on small holdings of less than 40 ha (100 acres); whereas, on the Westside, much of the land is owned by major corporations and holdings are much larger. The total irrigated area on the Westside is about one million hectares (2.3 million acres; Tanji 1990), of which 0.36 million hectares (0.89 million acres) are affected by salinity and sodicity, 0.25 million hectares (0.61 million acres) by a high water table, and 0.38 million hectares (0.93 million acres) by poor groundwater quality. Cotton is the major crop grown (more than 49 % of the irrigated area) on the Westside, with lesser areas planted to tomatoes and melons. Commodity prices and expected crop revenues have a significant impact on Westside cropping practices; a trend is developing toward growing more fruits, nuts and vegetables in preference to cotton. Subsurface tile drains have been installed on nearly 55,000 ha (135,000 acres) of irrigated lands (Salinity and Drainage Task Force 1992) to control seasonally high water tables and to dispose of salts flushed out from the root zone. A variety of methods are employed to manage irrigation, sustain soil fertility, and control crop pests: pre-planting irrigation, irrigation scheduling during the growing season, and crop rotations.

Administratively, the Westside is divided into water (or irrigation) and drainage districts. Irrigation districts are typically entities that deal with power as well as water resources. The Westlands Water District (WWD) is the largest water district on the Westside, and it has neither a natural drainage outlet nor an historic right to convey drainage water to the San Joaquin River. The San Luis Drain, with its terminus at Kesterson Reservoir, provided drainage relief to 2,150 ha (5,300 acres) of irrigated land within a 17,000 ha (42,000 acres) area in the northeast corner of the WWD until 1986. Because of selenium (Se) toxicity, discharge of subsurface drains water from these 2,150 ha (5,300 acres) of the WWD ceased in 1986.

Although the on-farm tile drains in the WWD that discharged through subsurface collector drains into the San Luis Drain were plugged in early 1986, on-farm subsurface drains in water districts north of Westlands in the Grasslands Basin continued to discharge salt- and selenium-contaminated drainage water to the San Joaquin River. These drains convey between 55.5×10^6 m^3 (45×10^3 acre-feet) and 92.5×10^6 m^3 (75×10^3 acre-feet) of combined surface and subsurface drainage water to the river annually, depending on annual precipitation and project water deliveries to agricultural contractors (San Joaquin Valley Drainage Program 1990). These effluents contribute to the gradual increase of Se loads in the San Francisco Bay. Therefore, it is necessary to evaluate the long-term changes in the Se content in the Bay and their relationship to the disposal of drain waters into the San Joaquin River.

Since the completion of the CVP's Delta-Mendota Canal, more than 18 million metric tons of salt have been imported into the Valley through water deliveries (Orlob 1991). Originating at the Tracy Pumping Plant on the San Joaquin River, water in the Delta-Mendota Canal typically has total dissolved solids (TDS) of 400–500 mg L^{-1}, while the TDS of California Aqueduct water supply is typically

250–350 mg L^{-1}. Average salt accretion to the San Joaquin Basin through the Delta Mendota Canal is estimated to be between 1.10 and 1.42 $\times$ 10^6 Mg (metric tons) per year (Orlob 1991).

The snowpack of the southern Sierra Nevada Mountains is the primary source of surface water flow in the San Joaquin Valley during the growing season, and most of the rivers and streams draining this mountain range have been dammed, beginning with the City of San Francisco's Hetch-Hetchy Dam and Reservoir, which provides for export of runoff to the San Francisco Bay Area. Since the construction of the surface water reservoirs on the eastside of the San Joaquin Valley and subsequent diversion of water along the Friant-Kern Canal, the average annual flow in the San Joaquin River (as measured at the Vernalis Gaging Station) has declined by about 16 $\times$ 10^9 m^3 (13 $\times$ 10^6 acre-feet).

Irrigation development and the construction of eastside reservoirs combined with the importation of salts in irrigation diversions from the Delta have decreased the frequency and magnitude of flood flows that once performed a useful purpose by flushing salt from the San Joaquin Basin. Salts continue to accumulate in the soil or in the groundwater aquifers beneath agricultural land on the Westside. The estimated net annual import for the period 1985–2005 has been modeled by Schoups et al. (2005), showing a steady cumulative increase in net salt in San Joaquin Valley soils and groundwater, confirming Orlob's (1991) salinity model projection of net salt load imports to the Valley. Because this long-term salt load imbalance will diminish irrigation sustainability, the pressure to implement mitigation measures has never been greater in California's history. To address the salinization, drainage, and drainage-related problems on the Westside of the San Joaquin Valley, the following measures have been investigated and, in many cases, implemented:

- Reduction of deep percolation (the downward movement of water below the root zone, past drains to the local groundwater system) through the adoption of water conserving irrigation technologies and practices, better irrigation scheduling, and changes in cropping practices.
- Reuse of drain water, through the use of salt-tolerant crops and agro-forestry.
- Manipulation of the water table to meet part of the crop evapotranspiration (ET) requirements.
- Conjunctive use of groundwater to meet a portion of crop needs.
- Improved instrumentation and monitoring systems to produce accurate and timely information and improve access to this information by growers.
- Development and installation of monitoring systems to evaluate progressive changes in soil and water quality in the terrestrial and aquatic ecosystems over time.

Prior to the unexpected crisis at Kesterson Reservoir in 1983, local, state, and federal policies had supported the development and expansion of agriculture in the San Joaquin Valley for more than a century. As consequences of expansion activities stemming from the Reclamation Act of 1902, which promoted land development for agriculture, municipalities, and industry, the wetland area decreased dramatically to less than 400,000 ha (a million acres) by the early

1920s and by 1977, wetlands acreage in California had dwindled to about 10 % of what it was in 1850, when California became a state of the USA (USGS 1996). During the middle of the nineteenth century, California boasted in excess of 1.6 million hectares (4 million acres) of wetlands (USGS 1996), presumably including estuarine wetlands, such as the Suisun marsh in the Delta.

Since the 1983 selenium crisis, public perception of agriculture has changed and policies favoring agriculture have been re-evaluated and modified, reflecting a re-balancing of competing resource management objectives. Federal water contracts within the CVP have been re-evaluated, under the Central Valley Project Improvement Act (CVPIA, PL 102-575, Title 34). The Environmental Impact Statement (EIS) requirement for contract renewal for each CVP contract may ultimately reduce the total allocation of federally developed water to agriculture. The CVPIA mandates that 986×10^6 m^3 (0.8×10^6 acre-feet) of water, once allocated to agriculture, be re-designated for fish and wildlife purposes. The CVPIA seeks to redress the loss of wetland acreage that has occurred during the past century. Other re-allocations of water supply from agriculture and urban uses to environmental uses, including several settlements of water rights disputes, have further reduced the water available to support irrigated agriculture in the San Joaquin Valley.

As the twenty-first century gets underway, agriculture on the Westside of the San Joaquin Valley is constrained by balancing the opposing objectives of maximizing agricultural production and on-farm income with the less tangible quality of life objectives stemming from prosperity. The vision of science and technology is that through a judicious management of water and land resources, California can continue to benefit from agriculture over many decades to come. This vision, nevertheless, has to be tempered by the recognition that policy decisions will need to weigh scientific solutions in conjunction with social objectives. In this context, it seems reasonable to assume that a sound grasp of the scientific and technological consequences of management alternatives is a prerequisite to help choose among competing social decisions. Consequently, science and technology professionals must appreciate their objectives in a broader context and provide information in a fashion that is amenable to social decision-making. Motivated by this perceived connection between science and technology on the one hand and social decisions on the other, we now assess our current knowledge of issues related to agriculture, drainage, and water management on the Westside (Quinn 2011).

3.4 The Framework for Scientific Research on the Westside of the San Joaquin Valley

3.4.1 The Linkage Water Provides

Scientific research needed to address the salinity/drainage issues generated by irrigated agricultural activities on the Westside is multi-scale. The research concerns of the Westside span a variety of scales in space and in time, addressing problems associated with (a) individual fields on a single farm, (b) multiple adjacent fields constituting the farm itself, (c) water and drainage districts that include a number of farms, and (d) the entire west side of the Valley. On the temporal scale, they include problems ranging from infiltration response and evaporation effects on a time-scale of hours, through irrigation scheduling during the growing season and crop rotations spanning a number of years, to long-term water delivery contracts and land retirement decisions over several tens of years (decades). On-farm decisions will, of necessity, depend on process-intensive small spatial scale questions, giving consideration to site-specific attributes. They will also be concerned with small and intermediate time scale issues, not exceeding a few years. On the next larger scale, water districts and managers are concerned with questions pertaining to sub-districts covering thousands to tens of thousands of hectares of land, with time scales larger than a single season. Although answers to these questions will be influenced by the variety of site-specific and intra-seasonal attributes of the farm scale, such variations will have to be aggregated and integrated into fewer parameters and objective functions, perhaps less process-intensive. At the largest scale are issues pertaining to a district and to inter-district linkages. The relevant time scale may vary from a season (e.g. managing water quality in the San Joaquin River) to several decades (contractual obligations, response to global climate change). Each of these scales has its own importance in the overall endeavor, and the attributes of these scales are interlinked dynamically in both directions. For the overall venture to be successful, information must flow freely in both directions between the scales. Therefore, it is a basic necessity to formulate a unified framework within which issues and attributes on all spatial and temporal scales are interlinked.

Water provides this linkage. It begins as precipitation and moves on the land surface as surface runoff or infiltrates to recharge aquifers. Water has the ability to dissolve and transport sediments, minerals and nutrients. It constitutes the universal link which binds together issues of irrigated agriculture and the environment on all spatial and temporal scales on the Westside. Therefore, we have used water as a unifying theme for system integration and as the basis to formulate the needed overall framework for system integration.

Early in the 1960s, the concept of regional groundwater motion was an important development in the field of hydrogeology because it formalized ideas concerning the dynamic behavior of water in groundwater basins (Toth 1963). The simple but yet profound basis of this conceptualization is that water, precipitating as rain or snow at higher topographic locations, moves more or less vertically downward into recharge areas. Then, driven by gravity and its own potential energy, it begins its travel towards a point of discharge, the ultimate discharge point being the ocean. Below the water table in the recharge area, the flow path shifts to a sub-horizontal aspect and groundwater travels laterally, subject to the nature of the geological formations, before encountering discharge areas. Below discharge areas, the flow paths become vertically upward. Between an area of recharge and one of discharge, the path length may be as short as a few hundred meters in a shallow groundwater system or as long as tens of kilometers in deep systems. On a vertical cross-section of a geologic system, shallow, intermediate, and deep groundwater subsystems will coexist, being separated by regions of stagnation. Careful study of the groundwater system on the Westside by previous workers (Belitz 1988, 1990; Davis and Coplen 1989) confirms the relevance of this conceptual model and its utility in understanding the role the groundwater system has played in both the evolution and current state of irrigation and drainage on the Westside. Concomitant with the recognition of three-dimensional, gravity-driven flow patterns in groundwater systems, it was also recognized by geochemists (Chebotarev 1955) that the geochemistry of groundwater is intimately influenced by its recharge-discharge relationships. In a broad way, waters from recharge areas tend to be oxidizing and richer in calcium and bicarbonate; whereas, discharge areas tend to be characterized by reducing conditions and enriched in sodium chloride.

3.4.2 The Regional Scale: Salts and Selenium

In the context of regional hydrogeology and geochemistry described above, the special environmental concerns that arose from the selenium toxicity problem at the Kesterson Reservoir in the early 1980s provided an impetus for several researchers from the United States Geological Survey (USGS) to study the regional geochemical factors governing salinity, as well as the distribution of trace elements in the groundwater system. Reconnaissance studies by Presser et al. (1991) have shown that selenium, which occurs mostly in a reduced elemental state (Se^0) in the marine shales, had been oxidized to more soluble selenate (SeO_4^{2-}) and selenite (SeO_3^{2-}) forms. From these observations, it was inferred that groundwater in the recharge areas is subjected to an oxidizing environment. In the distal parts of the fans, soluble selenate occurs with sulfate and other salts in evaporite deposits. These evaporite deposits are typical of groundwater discharge areas, where salts are moved to the land surface by upward-moving groundwater and subjected to evaporative concentrations.

As groundwater moves down-gradient into the Valley trough, its oxidation potential tends to decrease. Also, calcium (Ca) tends to precipitate gradually in the form of calcite ($CaCO_3$) and gypsum ($CaSO_4$) (Doneen 1967). Thus, groundwater in the Valley trough, in the vicinity of the San Joaquin River tends to be richer in sodium chloride (NaCl) and sodium sulfate (Na_2SO_4) and is characterized by lower oxidation potential, compared to upstream areas. For example, at a site near the Mendota Airport, oxidizing irrigation waters have displaced native groundwater to depths in excess of 16 m (50 ft) over a protracted period of time, and, prior to irrigation, the Sierran Sand aquifer was part of a regional discharge area, characterized by low oxidation potentials and enriched in dissolved iron and manganese (Dubrovsky et al. 1990). Also prior to irrigation, selenium had been concentrated by the evaporation process in the near-surface soils at this site over a long period of time. Due to vigorous pumping of groundwater for irrigation, the water table of Sierran aquifers, declined dramatically between the late 1950s and the middle 1970s. This water table decline presumably led to a seasonal downward movement of groundwater and the downward transport of salts and trace elements. As the displacement proceeded, mobile selenium was removed from the aqueous phase by reduction to Se^0 (elemental form) then sorbed and/or precipitated in the vicinity of the interface between the two waters of contrasting redox states. Deverel and Fujii (1987) provided evidence to show that selenium has been concentrated evaporatively with other salts in areas of regional groundwater discharge and that NaCl and Na_2SO_4 waters were displaced by waters rich in $CaCO_3$ and $CaSO_4$. These findings suggested that surface and ground water transients and redox conditions played a significant role in determining soil and groundwater chemistry in the near-surface soils and groundwater aquifers.

Hydrogen isotope concentrations in groundwater have been used as a marker to estimate the depth to which irrigation water percolated after the completion of the CVP facilities in the 1960s. These data have also been used to estimate the regional depth distribution of salts and trace elements in the semi-confined aquifer. Gilliom (1991) has suggested that, in the vicinity of the Panoche Creek alluvial fan, most salts and trace elements leached from the near-surface soils are contained in a zone between 10 m (30 ft) and 46 m (150 ft) below the land surface.

The chemical nature of soils on the Westside is known to be significantly related to regional geologic setting and groundwater chemistry. Southard et al. (1986) studied soil samples from eight transects extending from the Coast Ranges on the west to the San Joaquin on the east. They found that in the upper parts of the alluvial fans, selenium content in the soils decreased with depth, suggestive of the important role played by sediment transport on soil chemistry. In the medial and distal parts of the fans, selenium content increased with depth, presumably indicating that the element was being displaced downwards by irrigation water. The selenium concentration variation with depth also indicated that the rate of downward displacement decreased with depth as oxidation potential decreased, until reductive immobilization occurred in the vicinity of the interface with Sierran Sands of low redox potential. With regard to major chemical constituents, these authors found that sodium and chloride increased towards the river, an observation that is in conformity with the classical pattern of groundwater evolution in regional systems.

Doner et al. (1989) carried out a comparative study of archival soils collected in 1946 and contemporary soils on the Westside with special focus on selenium and arsenic. Because the arid climate, these soils were notably poor in organic matter and hence in a general state of oxidation. The soils derived from the Coast Ranges sediments were found to be richer in selenium and arsenic than those derived from the Sierran sediments in the axial trough of the Westside. The selenium in its oxidized form of selenate tended to be mobilized and leached from the soil. Therefore, contemporary soils in the irrigated areas were found to contain less soluble selenium than the non-irrigated soils. Arsenic concentrations in contemporary soils were found to exceed those in archival soils, suggesting a pesticide source.

Doneen (1967) carried out a study of sediments collected from bore holes at 19 different locations along the then-proposed alignment of the San Luis Drain. Samples were collected from depths varying from 16 m (50 ft) to 160 m (500 ft).

The samples from the axial trough of the Valley (derived from the Sierran sediments) were found to have lower salt content than those of the fans (derived from the Coast Range sediments). Virgin soils were found to contain more salts than irrigated soils, suggestive of active leaching. Depth profiles in the medial and upper parts of the fans were found to contain significant amounts of gypsum. Fujii et al. (1988) made similar findings, collectively studying soils and sediments from three locations, each with a drain system of different age (1.5, 6, and 15 years). From the available data, it was reasonable to infer that the relatively fine-grained nature of the Coast Range sediments (Laudon and Belitz 1991), combined with their origin in the chemical weathering of marine sediments and the oxidation state of the system, resulted in selenium being sorbed onto soil particles in the form of selenite.

The dependence of selenium geochemistry on the oxidation state has been established further by studies at the Kesterson Reservoir by Tokunaga et al. (1994). At Kesterson, pond-bottom sediments, rich in organic matter and submerged under water, are generally under anaerobic, reducing conditions. Experimental observations using synchrotron radiation by Tokunaga et al. (1994) have shown that under reducing conditions, selenium usually exists in the zero-valent elemental form in the soil matrix. Other scientists participating in the UC Salinity Drainage Task Force helped to develop a more detailed picture of the transformation and movement of selenium and other trace elements under local and regional conditions. These relationships serve as a basis for modifying on-farm and regional-scale management of these trace elements.

3.4.3 The Farm Scale: Crop-Yield, Salinity, and Irrigation

Agricultural productivity and profitability are important factors influencing on-farm decisions on the Westside of the San Joaquin Valley. Investments in innovative irrigation technologies, tile drainage and management practices are predicated on the sustainability of agricultural production and the willingness of banks and other lending institutions to take calculated risks on obtaining returns from these investments. Recognizing the importance of profitability to agricultural decision makers, research undertaken by scientists, engineers and agronomists on the Westside over the past several decades has focused on issues related to increasing crop production and soil fertility maintenance.

3.4.3.1 Root Zone Moisture and Salinity

Maintaining high crop yield requires management of moisture content, aeration, and salinity in the root zone. Except for some phreatophytes, most crops cannot tolerate waterlogging of the root zone for extended periods; rather, they require unsaturated conditions in which moisture and circulating air coexist in the root zone. Given such an environment, plants extract water and selected salts from the soil for their growth and sustenance. Although much remains to be understood about the mechanisms of uptake of water and nutrients by plant roots, it is generally believed that the uptake of water and salts by plant roots is driven by both physical and chemical conditions. In unsaturated soils, water is held in soil pores by capillary forces stemming from the affinity of water to bind to the surfaces of the soil particles. Capillary forces become progressively large with decreasing moisture content. Hence, to extract water from the soil, plants have to expend energy to overcome capillary forces and gravity. Moreover, when water in the root-zone has high salinity, osmotic potentials become a significant factor and plants also have to spend additional energy to extract fresh water. When they are forced to expend excessive energy because of high root zone salinity, plants become "stressed." When moisture content in the root zone is relatively low, water is held in small pores under high capillary pressures, requiring plants to expend large amounts of energy to obtain the water necessary to meet transpiration needs and retain the fluid pressure inside a plant cell, which controls swelling and contraction of the cells (turgor pressure). When turgor pressure declines below a critical level, plants wilt – the most visible early signs of plant stress. To achieve maximum crop yields, water content in the root zone over the entire crop growing season should be neither too high to impair oxygen circulation nor too low to require high expenditures of energy to extract water. Salinity in the root zone must not rise above a certain, crop-specific, threshold level for an extended period of time. Certain crops, such as cotton, wheat and sorghum, exhibit greater tolerance to salinity than shallower-rooted vegetable crops, such as melons and tomatoes.

The volume of water required for crop establishment and growth is largely determined by the crop's ET requirement, the volume of water lost to the atmosphere from plant leaf transpiration and direct evaporation from the soil surface. ET varies from crop to crop and is also affected by additional factors, such as stage of growth, local climate, soil salinity, water table depth, and wind. Based on years of field experience (Westlands Water District 1984), the ET requirement has been shown to vary from less than 0.3 m (1 ft) for melons and peas to as much as 1.3 m (4.2 ft) for alfalfa (hay).

Many field-scales, multi-year experiments have contributed to the current knowledge of plant tolerance to stress and salinity. Although most plants do not grow optimally under stress, some specific plants, such as cotton, may withstand stress without adverse effects on yield during the later stages of the growing cycle (Ayars et al. 1990). Cotton lint yields can benefit from stress applied at critical times during the growing season, which helps to stimulate cotton boll production relative to plant biomass.

Because of the importance of moisture and salt regime in the root zone, understanding the physics of processes affecting these factors is important in the determination of crop yield. The currently used basis for such research is the Richards equation (Richards 1931), which describes the transient movement of moisture in unsaturated soils. The solution of this difficult-to-solve equation has been greatly facilitated in recent times with the advent of the computer. Many researchers now use mathematical models to study the dynamics of flow in the root zone, giving consideration, to the dependence of hydraulic conductivity and soil-moisture-capacitance on the moisture potential, as well as other factors. Among the early workers in this area of research are Hanks et al. (1969) and Bresler and Hanks (1969). Using appropriate forcing functions, such as rainfall, irrigation, evapotranspiration, and the fluctuation of the water table, models developed by these researchers sought to quantify the variations in moisture content and water fluxes in and around the root zone. In turn, the moisture content and water flux could be used to analyze the dynamic transport of salts and dissolved constituents as governed by advection, hydrodynamic dispersion, and molecular diffusion (Freeze and Cherry 1979).

A fundamental attribute of the Richards equation is that it is a mass-balance equation for water, incorporating both evapotranspiration and plant uptake of water, which function as external boundary conditions for the equation. Therefore, data pertaining to these conditions have to be generated from independent experiments or models in order to analyze water balance in the root zone. Methods based on micrometeorological data have been developed to estimate evaporation and soil hydraulic diffusivity on the field scale (Parlange et al. 1993; Katul et al. 1993). Based on a detailed study of bare soil evaporation from salt-encrusted soils at the Kesterson Reservoir, Zawislanski et al. (1991) concluded that the evaporation rate in such a soil is usually small, being dominated by vapor transport.

For on-farm design and planning, simple empirical models of evaporation and transpiration are often used to make design and management decisions relating to crop irrigation scheduling and crop production. Irrigation scheduling software, such as ROY; SWAP-ET and the expert system AGWATER, give consideration to evaporation and transpiration in irrigation scheduling. Because the extraction of water from the soil by plant roots is critical, agronomists have devoted attention to developing techniques for quantifying uptake of water by plants. The interchange of water between the plant root and the soil can be considered, as a first approximation, to be dictated by two different driving forces. The first, predominant force, which is hydromechanical in nature, drives water from the soil into roots when the hydraulic potential (or, hydraulic head) within the roots is lower than that in the soil. The hydraulic head includes components due to gravity as well as matric potential (or, moisture suction). The uptake in this case is inversely proportional to the root hydraulic resistance arising from root geometry, the biomechanical properties of the root tissue, and the hydraulic resistance of the soil in the vicinity of the root. The second force arises from osmotic processes when the soil water and the cells within the plant root differ in solute concentration. The root is known to behave as a semi-permeable membrane, allowing the free flow of water but preventing the movement of relatively large dissolved ions. The nature of the osmotic gradient is such that fresh water tends to move in the direction of increasing salinity across a semi-permeable membrane. As the soil water salinity increases, increasing osmotic forces will tend to reduce the movement of fresh water from the soil to the root. At very high soil-water salinity, this flux is reduced to near zero or even reversed, which can prevent the plant from extracting adequate moisture from the soil to replace transpiration losses, leading to permanent wilting (the state of plant stress which does not allow recovery). When the soil moisture content is low and soil salinity is relatively high, the osmotic gradient between the plant and the soil solution may be significant. In this case, both the hydromechanical and osmotic forces should be given consideration in estimating root uptake of water. Gardner (1960) suggested the use of a mechanistic model for plant water uptake, restricting attention to the difference in hydraulic head between the soil and the plant root. This line of reasoning has been followed by other researchers, including Nimah and Hanks (1973).

Experience with crop yield in saline lands has shown that crop yield is significantly related to soil-water salinity in the root zone (van Genuchten and Hoffman 1984). To address this issue, van Genuchten (1987) proposed a model that accounts simultaneously for the effects of hydraulic and osmotic forces. Cardon and Letey (1992a) evaluated these two types of approaches by incorporating them into a numerical model and applying the model to field data. They found that under saline conditions, the purely mechanistic model of plant water uptake may not be sufficiently reliable because of other important factors influencing plant physiology.

Closely related to plant-water uptake is the crop yield function, which is used extensively in models dealing with economics and management. The maximum yield, Y_{max}, is the crop yield that may be expected under ideal conditions of adequate supplies of non-saline water. Under less-than-ideal conditions of reduced water supply or increased salinity or both, the yield will be less than Y_{max}. Different crop-yield functions have been proposed in the literature, as discussed by Letey and Knapp (1990).

3.4.3.2 Leaching the Root Zone

Maintaining optimal salinity in the root zone necessarily implies that excess salt must be removed from the root zone. Irrigation engineers have long recognized the need to provide a certain amount of "excess" water to flush accumulated salts from the root zone. Here the word "excess" refers to the amount of water exceeding the ET requirements of the plant. Two important parameters, "leaching required" and "leaching achieved", are used quantitatively by irrigation engineers. Hoffman (1990) provides a summary discussion of these concepts.

On the Westside, excess salinity in the root zone poses a particularly challenging problem requiring deliberate management interventions. A common strategy of salinity management is to pre-irrigate the field during the winter or early spring months to flush salts from the root zone. Growers with unused water supply allocations at the end of the water contracting year have also been known to apply large volumes of water to attempt to "bank" water in the shallow groundwater aquifer for the following irrigation season. In the western San Joaquin Valley, dispersion of fine particles and surface sealing of soils decrease infiltration capacity of soils and make it difficult to meet crop ET needs in the latter part of the irrigation season. Obviously, practices such as pre-irrigation and (water) banking, abetted institutionally by the timing of the water contracting year, have exacerbated saline shallow groundwater and drainage problems on the Westside. It has been estimated (Salinity and Drainage Task Force 1992) that deep percolation arising from pre-season irrigation applications alone may amount to 0.15–0.3 m (0.50–1 ft) of water per year.

Mathematical models commonly used by irrigation engineers to generate quantitative understanding of leaching from the root zone are based on a mass-balance of water as well as salt in a one-dimensional column of soil, extending from the soil surface through the root zone to the water table. For water balance, these models include rainfall, applied irrigation water, evaporation and transpiration, and changes in stored moisture as a function of time. The change in the amount of salt in the root zone as a function of time includes the amount of salt imported with the irrigation water, precipitation and dissolution of minerals, salts transported by deep percolation to the water table, and the salts removed with the harvested crop. Although conceptually simple, quantified estimation of the various components factoring into the equations of mass-balance is extremely difficult. As a result, practical problems of salinity management in the field are solved with approximations. An example of such idealized models is that of Hoffman and van Genuchten (1983), which uses a linearly averaged salt concentration for the root zone. Letey et al. (2011) demonstrated that in comparison to the transient state approach, salt leaching requirements established on the basis of steady state water and salt flows, such as in Hoffman and van Genuchten (1983) and in Ayers and Westcot (1985), overestimated salt buildup in the soil profile and thus the harmful consequences of irrigation. The discrepancies are especially large at low leaching fractions.

3.4.3.3 Irrigation Efficiency

Closely related to the concept of leaching requirement is that of irrigation efficiency. Irrigation efficiency has been defined in many different ways for both practical and policy reasons. One definition of irrigation efficiency is the ratio of the irrigation water applied to the crop to the amount of water beneficially used by the crop. Beneficial use is commonly defined as the annual crop ET or crop water requirement. Some irrigation consultants and agricultural water districts have a broader definition of crop beneficial use, which includes a minimum leaching requirement and an allowance for certain cropping practices, such as frost protection. The addition of these other factors produces higher estimates of irrigation efficiency, since beneficial use is the denominator in the equation. Irrigation efficiencies greater than 100 % have been reported by Westlands Water District in areas where shallow groundwater is utilized by the crop to satisfy a portion of crop ET. In such areas, the irrigation efficiency, if computed for each irrigation event, shows an increase over the irrigation season as crop roots develop and become extensive enough to intercept capillary water from the shallow groundwater table. Calculated irrigation efficiencies are typically lower for individual irrigation events than for the irrigation season as a whole. It follows therefore that depending on whether one considers a single furrow, a single farm, or a whole water district, or whether one considers a single irrigation event, irrigation over an entire season, or irrigation over the calendar year, irrigation efficiency is a concept subject to spatial as well as temporal scale variations.

Maximizing water use efficiency may not be in the best interest of the grower in circumstances where soils are heterogeneous in their hydraulic properties and where it is difficult to obtain high distribution uniformity in irrigation applications. Burt et al. (1992) has suggested a practical maximum distribution uniformity of 80 % for most furrow irrigation systems. With poor distribution uniformity, high irrigation water use efficiency is not possible, if all parts of the field are to be provided with water sufficient to meet crop ET requirements. In some circumstances, such as those that exist in the western San Joaquin Valley, where selenium-contaminated drainage is produced in proportion to the excess irrigation applied to the crop, it may become more cost effective to maximize water use efficiency and minimize drainage discharge. Hatchett et al. (1989) demonstrated this strategy, using the Westside Agricultural Drainage Economics Model (WADE) in which increasing costs of drainage disposal led to reductions in applied irrigation water. In these circumstances, some portion of the field receives less than an optimal water supply, thereby reducing the evapotranspiration of plants, which, in turn, leads to a reduction in crop yield. During a drought in 1992, water deliveries to growers were reduced by as much as 75 %, which led to a large reduction in irrigated acreage. Water District figures for average crop irrigation applications during the drought period appear to support conclusions drawn from the WADE model analysis.

Salts, flushed from the root zone, are displaced downwards to the water table. The migrating salts displace and mix with resident groundwater and generally act to degrade water quality at the interface between the saturated and vadose zone. Refluxing of shallow groundwater by evaporative concentration and irrigation flushing can increase the salinity of vadose zone water throughout the irrigation season. Where drains are present, some portion of the saline percolating water is intercepted by the drains and discharged to sumps or surface drainage ditches. Should strong lateral groundwater flow exist below the water table, groundwater salinity may not change significantly because percolating water would continually dilute and displace resident saline water, which in turn would migrate and disperse in the direction of the groundwater gradient. However, in the vicinity of the Valley trough where vertical groundwater gradients are minimal and lateral groundwater motion is sluggish, salt concentrations in the shallow groundwater aquifer can build up to high levels, owing to frequent refluxing before the water passes below an extinction depth beyond which the effect of evaporative concentration declines to zero. Shallow groundwater salinity can achieve concentrations of between 20 and 30 times the applied water salinity.

Because groundwater is a valuable natural resource in itself, concerns are being raised about the long-term viability of the groundwater aquifer as a source of good quality irrigation water. Deep percolation of irrigation water displaces better quality groundwater and the rate at which this occurs in the upper aquifer is a function of groundwater pumping in the deep semi-confined and lower confined aquifers, which in turn affects the vertical hydraulic gradient in the upper aquifer. To address concerns about irrigation-induced salinization of the aquifers, it is necessary to address agronomic practices, such as over-irrigation and groundwater pumping. The first step towards achieving this end consists of reducing irrigation deep percolation by minimizing the leaching requirement through effective irrigation scheduling, making prudent choices in irrigation technology, and tailoring crop ET needs. In addition, growers may manage to maintain salt levels as high as possible in the root zone as will be tolerated by the plants. Letey and Knapp (1990) provide an account of how dynamic programming methods can help in designing irrigation schedules based on management of irrigation efficiency and distribution uniformity. Both these measures of performance can be affected by the choice of irrigation technology as well as irrigation water management. On the Westside, a variety of water application methods is used, including furrow, corrugation, basin, and border irrigation, as well as micro-irrigation methods comprising surface- and subsurface drip irrigation and subirrigation (Kruse et al. 1990). Sprinkler irrigation systems in use can be divided into permanent, moved, side roll and center pivot. Burt et al. (1992) made the point that without deliberate attempts to improve irrigation management, adoption of new technologies by themselves might do little to improve irrigation water use efficiency. Ayars et al. (1987) described strategies to improve irrigation water use efficiency: improved irrigation methods, reduction of pre-planting water application, reduction of irrigation applications as crops near maturity, partial re-use of drain water, and use of on-farm indicators to guide dynamic water application.

3.4.3.4 Water Table Management and Subsurface Drains

Plant roots may draw water either from above (from applied irrigation water) or from below (upward capillary flow from the water table). Consequently, shallow water tables can be an important source of water to satisfy crop ET needs. Many researchers have recognized the potential of the water table to supply water to the root zone and have investigated the mechanisms involved, both theoretically and experimentally. One of the goals of these workers has been to estimate the component of the crop ET needs supplied by the water table. Early work in this regard was based on steady-state unsaturated flow idealization (Gardner 1957). Based on carefully controlled field experiments over a 3-year period in North Dakota, Benz et al. (1981) found that maximum crop productivity was obtained when the water table was at an optimum depth of 1–1.5 m (3–4.5 ft). Water tables that were too shallow led to waterlogging of plant roots and limited the depth of root penetration. Grimes and Henderson (1986) found that in areas of shallow water tables, 50–60 % of the crop ET could be supplied below the root zone directly from the water table. The actual uptake by the plants was found by these authors to depend on the salinity and depth of near-surface groundwater. These authors also found that in order to manage the water table efficiently, as a source of crop ET needs, site-specific monitoring and data collection are necessary, and a tile drain system should be in place to allow some control of upward capillary flow.

A consequence of shutting down the San Luis Drain, accompanied by plugging the drains of the WWD in 1986, was to force affected growers to search for on-farm solutions to manage drainage without any means of disposal outside the district. One strategy that has resulted from this search for in-Valley solutions is the

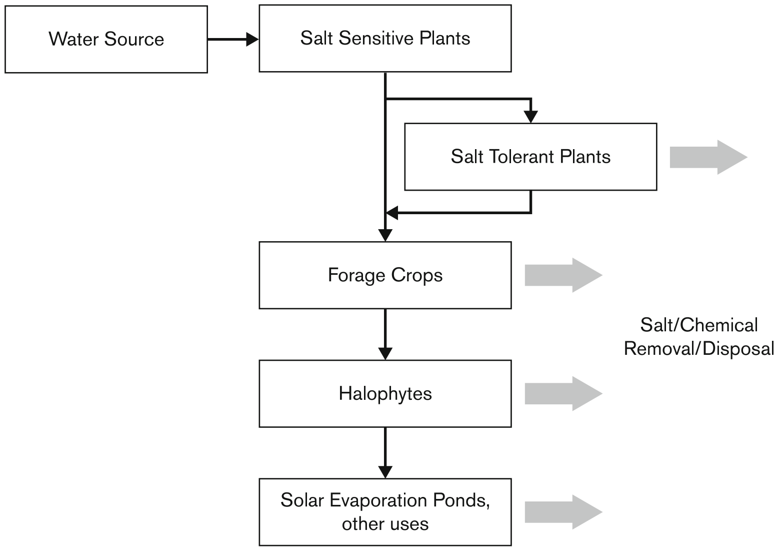

Fig. 3.3 Successive reuses of irrigation water and agroforestry crops within the Integrated On-Farm Drainage Management (IFDM) System

sequential use of progressively salt-enriched drainage waters and the final disposal of unusable brine into solar salt ponds, as recommended by the San Joaquin Valley Drainage Program (1990). John Deiner, landowner at Red Rock Ranch, the Department of Water Resources and Cervinka (1990) have been conducting field experiments on a sequential system in which subsurface drainage water from salt-sensitive crops is used to irrigate salt-tolerant crops, followed by salt-resistant trees, such as eucalyptus, and finally, by halophytes , plants which thrive on saline environment (Fig. 3.3). Although these salt-resistant trees help to lower the water table over the first few years, the gradual buildup of salts inevitably follows, diminishing the abilities of these plants to extract water from the saline soil environment (Tanji and Karajeh 1993). Tail water from the halophytes is fed into small, lined experimental evaporation ponds to recover salts, primarily sodium sulfate, which has some commercial value. If alternate beneficial uses could be found for large quantities of salt residues from evaporation ponds, then lined evaporation ponds may be able to recover their costs of construction with the added benefit that they would prevent contamination of the local groundwater system. Researchers within the UC Salinity Drainage Task Force have continued to advance this concept at the Red Rock Ranch site. This concept is now known as Integrated On-Farm Drainage Management (IFDM).

At the farm level, the design and depth of the drain as well as the spacing are ideally intended to obtain the greatest drainage yield at minimum cost. In practice, installation of drains is limited by the availability of equipment and the grade in the field. During the past decade, researchers on the Westside have recognized that the lateral flow of water into drains from the groundwater system must be considered in the design of drains (Grismer 1993). That the drains interact with the shallow groundwater system on the Westside has been documented by Deverel and Fujii (1987), Deverel and Fio (1991), and Fio and Deverel (1991). More recent cooperative research between the US Bureau of Reclamation (USBR) and the Advanced Decision Support System Group at Colorado State University (Quinn 1993) has explored the effect of retrofitting existing drainage systems to minimize interception of groundwater high in selenium and boron. Where groundwater shows increasing salt and trace element concentrations with depth, shallower, closely spaced drains produce drainage with lower concentrations of salts, selenium, and boron. Tailoring the retrofit design to the contaminant depth distribution characteristics in the shallow aquifer at affordable cost may be a viable alternative for irrigated cropland in the Grasslands Basin and also in the WWD. In the Westlands case, minimizing the trace element loading and drainage flow would help to reduce the cost of drainage treatment and disposal.

In a project to aid the characterization of shallow aquifer hydrogeochemistry in the Grasslands Basin, Ayars and Meek (1994) statistically analyzed water flow and salt load data from the drainage system of the Panoche Drainage District for the period 1986–1991. The analysis showed a strong correlation between drainage flow and salt load, but this relationship tended to be unique for individual drainage sumps. The proportion of deep percolation and the proportion of shallow and deep groundwater that were intercepted at each site differed. The apparent stability of this relationship would imply that the majority of tile drainage was displaced groundwater, since deep percolation is more likely to have variable concentration over the irrigation season. Further studies to characterize the drainage systems in the remainder of the Grasslands Basin will be necessary to allow regional management of drainage contaminant loading to the San Joaquin River. These research studies are likely to proceed, using a hierarchical systems approach, integrating field and modeling studies at the field, district and regional scales.

3.4.3.5 Site Characterization

Estimating the hydraulic conductivity and soil-moisture capacity and their dependence on soil moisture suction is commonly determined by laboratory tests on soil core samples collected in the field. Traditionally, soil hydraulic conductivity measurements have been made using steady-state flow experiments. Recent research (e.g. Eching and Hopmans 1993; Eching et al. 1994) has tended towards the use of transient experiments, which take into account soil moisture capacitance. In addition, efforts are also made to estimate the hydraulic diffusivity (that is, hydraulic conductivity divided by field soil moisture capacity) on a field scale. Parlange et al. (1993) reports the development of a technique to this end, based on micrometeorological data.

The greatest challenge to site characterization stems from the spatial variability of the natural soil, an unavoidable impediment to the confident application of the principles of physical science to the solution of practical problems related to agriculture and irrigation on the Westside. Loams, silts, sands and clays in the form of stringers, lenses, and layers occur in partly disordered distribution that profoundly influences the patterns of water and solute movement within the soil system. As a consequence of spatial variability, these patterns are difficult to comprehend, characterize, or incorporate into management. Nielsen et al. (1973) and Biggar and Nielsen (1976) carried out detailed field investigations to understand the spatial variability of hydraulic properties, solute displacement, and pore water velocities over a 150 ha (371 acres) field plot. These early studies suggested that field-scale pore water velocities are log-normally distributed. These studies also provided insights into the number of field samples that need to be collected to generate estimates within a prescribed accuracy and showed that the number of samples needed depends on the nature and extent of spatial variability.

The dynamic processes of flow and transport can be interpreted in terms of the Richards equation and the advective-dispersive equation, provided that adequate process-relevant data are available on a scale with sufficient fine-tuning. Detailed characterization of soil features on a scale of a meter or less within a system with dimensions of hundreds of meters is too expensive and not warranted. Therefore, logically consistent and efficient methods must be found by which the effects of small-scale heterogeneity can be incorporated into the larger scale idealizations so that practical problems of farm-level management can be solved beneficially. This challenge has led to two different approaches of problem solving. Some researchers consider the problem of spatial variability to be a formidable one in relation to the resources that can be reasonably invested for their resolution. Accordingly, they favor using simple, empirical functions for problem solving. One such approach is the transfer function method (Jury et al. 1986; Sposito et al. 1986). In this approach, simple empirical functions connecting cause and effect are used to characterize the system, without giving consideration to process details. Clearly, these functions will be site-specific. Other researchers, seeking to be guided by detailed process-oriented models, choose to use stochastic theory to scale up hydraulic parameters from a smaller scale to larger scales (Hopmans et al. 1991; Clausnitzer et al. 1992). Childs et al. (1993) carefully measured infiltration rates at many points over a 21 ha (52 acres) plot irrigated by furrows, using a newly designed furrow infiltrometer. Statistical analysis of the spatial and temporal variations of measured infiltration showed that mean values and variability of irrigation rates could be estimated with data from representative sites, rather than resorting to complete field interrogation. Effective management of salinity and drainage in a regulatory environment that precludes a drain to the ocean requires practical solutions to the issue of spatial heterogeneity.

3.4.3.6 Real-Time Monitoring and Instrumentation

The San Joaquin River Management Program Subcommittee on Water Quality, an interagency group including personnel from Lawrence Berkeley National Laboratory, the California Department of Water Resources (DWR), the California

Regional Water Quality Control Board (RWQCB) and the US Geological Survey (USGS), embarked on a study to demonstrate the utility of real-time water quality management on the San Joaquin River. The initial phase of the project required the upgrading of monitoring stations in the Panoche Water District, in Mud Slough and Salt Slough and along the main stem of the San Joaquin River to allow continuous reporting of electrical conductance (EC), temperature and flow. Radio telemetry was used to link each of the monitoring stations with a central computer, as well as allowing data access to authorized users. Associated cooperative research within the UC Salinity Drainage Task Force that involved Panoche Water District and the University of Rhode Island attempted to develop on-farm and district-level strategies to regulate the flow of drainage and the selenium and boron loads in drainage from the Panoche Water District and surrounding agricultural water districts. These quick-response strategies have involved operation of a combination of facilities, such as district temporary holding ponds, district drainage recirculation, and recycling pumps (Quinn and Karkoski 1998; Quinn 2009a). Water conservation and drainage recycling programs have further enhanced the ability of the district to control the magnitude and timing of salt and selenium drainage exports.

More recent research has investigated seasonal wetland salinity management and the impacts of manipulating drainage scheduling on wetland habitat sustainability (Quinn 2009b; Quinn and Hanna 2003). This research was conducted on both the regional scale and at the scale of individual wetland impoundments and produced the first vegetation change detection maps derived from high resolution remote sensing in the watershed for managed wetlands in the state, federal, and private wetland areas.

Several of these drainage management projects by members of the UC Salinity Drainage Task Force have helped to integrate knowledge of surface and subsurface hydrological processes from the farm to the regional scale within a single decision support system. To many hydrologists, the major challenge for the next decade is the creation of decision support systems which integrate, in a defensible manner, our current best thinking and best mathematical representations of surface and subsurface hydrological systems.

3.5 The Status of Mathematical Models

During the past two and half decades, considerable research has occurred on the irrigation and drainage hydrology of the western San Joaquin Valley, ranging from micro-level laboratory scale studies to macro-level river basin planning studies. Of these studies, few have been successful at modeling the surface and subsurface hydrologic systems with equal rigor. With the large increases in numerical processing power in the past decade, this failing has less to do with computing resources and more to do with conceptual limitations, disciplinary bias, and short-sightedness. For example, hydrogeologists rarely consider irrigation technologies and water management practices in the application of groundwater models; likewise, irrigation engineers rarely consider the groundwater aquifer in their models much below the bottom of the crop root zone. An integrative approach is needed, since it is certain that models will assume an increased role in river basin planning activities in the future. Models can be divided into several groups: process-oriented models, optimization models, planning models, and expert systems.

3.5.1 Process-Oriented Models

Process-oriented models seek to analyze the interaction of fluid motion with plant roots and the atmosphere, the transport of heat and dissolved chemical components within the soil, and the chemical interactions that invariably occur between the aqueous phase and the solid phase. Chemical reactions between mixing waters and between the aqueous and solid phases are governed by the equations of chemical thermodynamics. Because water ultimately governs the transport of chemical constituents and heat, modeling of water flow is fundamental to all of these models. The equation governing the transient flow of water in unsaturated-saturated, deformable media constitutes the basis for solving solute transport problems, taking into account advective, dispersive, and diffusive processes. Dynamic process models combining fluid flow and solute transport have been applied to the study of local, farm-level problems, as well as regional-scale problems on the west side of the San Joaquin Valley.

3.5.1.1 Regional Process Modeling

To gain an understanding of the regional groundwater flow conditions in the central part of the western San Joaquin Valley for the purpose of developing drainage management strategies, Belitz et al. (1993) carried out three-dimensional, transient groundwater flow simulations using a mathematical model. The simulated area covered 1,410 km^2 (551 mile^2), comprising 11 water districts. The semi-confined zone overlying the Corcoran Clay was divided vertically into five layers, of which the top two layers were assigned constant thicknesses, while the remaining three were assigned spatially variable thicknesses. The confined aquifer was treated as a single layer. The low-permeability Corcoran Clay was represented through a "leakage" term governing the water transfer between the semi-confined zone and the confined aquifer. The simulations considered aerially variable recharge rates to account for irrigation effects, pumping from wells, effects of subsurface drain systems and bare soil evaporation. In the horizontal plane, each grid block had an area of 2.56 km^2 (1 mi^2). The hydraulic properties of the materials (hydraulic conductivity, specific storage, porosity and specific yield) were initially assigned on the basis of lithology, data from laboratory tests, and data gathered from hydraulic tests conducted in the field. Later, these values were fine-tuned through the model calibration process, using field data gathered from specific sites between 1972 and 1984. Having calibrated the model, Belitz and Phillips (1993, 1995) then applied the model to study the response of the system to alternate strategies of irrigation and drainage management. One finding of this study was that vertical flow predominates in the regional flow system. Based on this result, the authors inferred deliberate land idling or retirement would have a noticeable impact only in the immediate vicinity of the land. This inference is at variance with the commonly held belief that upslope irrigators are contributors to down-slope drainage problems.

Belitz and Phillips (1993, 1995) examined alternate management strategies that have occurred and that could occur during the 50-year period, 1990–2040. If the status quo were to continue, they inferred that the area experiencing bare-soil evaporation would expand from about 575 km^2 (224 $mile^2$) to about 880 km^2 (344 $mile^2$), and the flow in the drains would increase from about 3,100 hectare-meter (25,000 acre-feet) to about 3,450 ha-m (28,000 acre-feet). Alternate simulations suggested that by reducing irrigation recharge from 40 to 15 % and by gradually increasing groundwater pumping, the area under bare soil evaporation could be reduced to about 200 km^2 (78 $mile^2$) and drainage flow could be reduced to about 10×10^6 m^3 (8×10^3 acre-feet). The authors estimated that 50×10^6 m^3 (0.4×10^6 acre-feet) of water could be released for other beneficial uses.

Fio (1994), using the regional model of Belitz and Phillips (1993, 1995), carried out sub-regional scale simulation of groundwater flow and drain interactions over an area largely defined by the boundaries of Panoche Water District. Results generated from the regional model provided a basis for specifying the boundary conditions for these simulations. Attention was restricted to the upper 26 m (85 ft) of the semi-confined zone. This model generally confirmed the results of the regional model with regard to lateral flow but found instances in which vertical flow estimates of the regional model were at variance with field data.

3.5.1.2 Farm-Scale Modeling

On the local scale, process-oriented mathematical models have been used for a variety of purposes, including the study of root zone water movement, water table fluctuations, evapotranspiration, irrigation efficiency, and performance of drain systems. To study the dynamic interactions between plants and the water table, Cardon and Letey (1992b) evaluated two types of plant-water uptake equations proposed in the literature (Nimah and Hanks 1973; van Genuchten 1987) and concluded that under saline conditions, typical of many areas on the Westside, plant uptake models must give consideration to mechanical potentials (Darcy flow) as well as osmotic pressure variations, arising from spatial changes in salinity. Based on this finding, they coupled a form of van Genuchten's (1987) plant-water uptake model with one-dimensional transient water and solute transport model to analyze crop yield under dynamic water flow conditions. The credibility of this model was tested by supplying it with detailed data from a well-controlled greenhouse experiment (Cardon and Letey 1992c).

The efficiency of irrigation at the farm-level is dependent both on the method of irrigation (furrow or sprinkler irrigation) and the uniformity with which water is applied over the field. Using mathematical models restricted to saturated flow conditions, Ben-Asher and Ayars (1990) analyzed the effect of spatial non-uniformity in water application on deep seepage. An important issue addressed in the research was maximization of crop yield, since non-uniform irrigation causes deep percolation in some parts of the field and under-irrigation in others. Ben Asher and Ayars (1990) found that more deep percolation resulted from non-uniform sprinkler irrigation than from uniform sprinkler applications.

Selenium data collected by Deverel and Fio (1991) in their study of the sources of drainage flow showed that upward-moving groundwater flow from deep aquifer contributed more to the annual selenium load to the particular drain lateral (332 kg or 732 lb) when compared to flow from the root zone above (68 kg or 150 lb). Selenium captured from the deepest aquifer originated from water that had deep percolated prior to installation of the drains. Grismer (1989, 1993) has shown that "lateral flow" (or regional groundwater flow) is dynamically linked with the flow systems of the drains.

3.5.1.3 Reactive Chemical Transport Models

Although many of the process-oriented modeling efforts have been devoted to studying moisture movement and bulk salinity, researchers have recognized the importance of understanding chemical reactions that occur among the numerous chemical species within an irrigated system. These interactions not only involve the mixing of waters of contrasting chemical quality (such as the imported irrigation water, the water in the root zone, and the groundwater) but also the interactions between the water and the solid phases of the soils and the sediments. Chemical interactions are further modified by evaporative concentration and dilution due to infiltration. These reactions result in profound modifications of the composition of the aqueous phase and the solid phase through precipitation, dissolution, adsorption, and desorption. The quantitative analysis of these interactions involving several chemical species constitutes a computational complexity that is far more intensive than the task of modeling moisture movement. Therefore, the application of reactive chemical transport models to the resolution of problems on the Westside has become possible only within the past few years, as a new generation of computer work-stations have become available.

The reactive chemical transport models needed for understanding the behavior of major ions (e.g. Ca, Mg, Na, K, Al, Fe, Si, SO_4, Cl^-, HCO_3) and trace elements (e.g. As, Mo, Se, B) are characterized by two attributes. The first is the chemical transformation of the species into various forms (valence states, complexes and minerals) and their addition or removal from the aqueous phase. The second is the transport of the multitude of chemical species by the flowing water. In addition, the chemical transformations are also influenced in a major way by mobile gases (especially in the vadose zone), notably carbon dioxide and oxygen. The chemical transformations include redox reactions, hydrolysis, sorption and ion exchange.

These processes are interlinked by the constraints of chemical thermodynamics, such as electrical neutrality, the balance between electron donors and receptors, and mass conservation. The transport of chemical species, on the other hand, involves processes of advection, molecular diffusion and hydrodynamic dispersion. The task of reactive chemical transport modeling, therefore, entails the dynamic coupling of the transport modules with the various chemical reaction modules.

A quantitative understanding of the interactions between soil minerals and soil water is of fundamental importance for proper agricultural management. Therefore, even with the introduction of rudimentary computers in the early 1960s, soil scientists took the first steps toward applying the principles of equilibrium thermodynamics to interpret reactive chemical transport experiments on soils. Dutt (1962) and Dutt and Tanji (1962) pioneered the practical application of mathematical models for reactive chemical transport when they developed and validated a model for gypsum precipitation in the presence of exchangeable calcium and magnesium. Later, Tanji et al. (1972) applied a similar model to the study of the problem of land reclamation in the San Joaquin Valley. This model simulated, in a credible fashion, the effects of gypsum amendment on sodic soils at eight experimental field plots.

The availability of powerful computer workstations has encouraged the development of a number of generic reactive chemical transport models for simulating the simultaneous migration of several chemical components in the presence of fluid-solid interactions involving redox reactions, ion-exchange processes, and chemical kinetics. Among the generic models that have been available for analyzing fluid-solid interactions are Felmy et al. (1984), Wolery (1979), Parkhurst et al. (1980) and Mattigod and Sposito (1979).

Simunek and Suarez (1993) have focused attention on modeling the role of carbon dioxide in controlling vadose zone chemistry and hence, the yield of crops on irrigated lands. Their model couples the temperature-dependent production of carbon dioxide due to microbial action and root respiration and its subsequent transport by transient moisture movement in the unsaturated zone. Suarez and Simunek (1993) applied the model to data from field experiments and found reasonable agreement, thereby supporting the model's credibility.

3.5.2 Optimization Models

Although process-based models can help to evaluate how crop yield and salt-load generation will respond to various strategies of water application and drainage, activities such as leaching, aimed at increasing crop yield, may conflict with protection of groundwater quality in the shallow groundwater aquifer in the long term. In addition, these competing objectives are also constrained by "real-life" economic considerations of cost and benefit, which are not accounted for in process models but are addressed in optimization models. To help make optimal decisions under "field" conditions, many researchers have applied linear and dynamic programming methods (Knapp and Wichelns 1990). The decisions involved may apply to a single farm or group of farms and are intended to lead to identifying the optimal irrigation strategy, giving consideration to crop needs, salinity and the cost of production.

Optimization models constitute a primary interface between science-based resource assessment on the one hand and resource economics on the other. Agricultural resource economics and associated policy development constitute a very active field of inquiry in the social sciences (e.g. Carlson et al. 1993). Using as their basis the physical attributes of resource systems, such as soil and water, researchers involved with economics and policy development investigate how these resources will respond to various strategies of resource utilization and social behavior. Based on such analyses, they proceed to understand how, through various strategies of marketing, regulation, incentives and taxation, widespread social behavior(s) can be modified towards a sustained utilization of natural resources. In this context, optimization models constitute an analytical tool capable of quantifying the complex interactions of a multitude of variables.

At the farm level, the ideal optimization model would consider several alternative strategies that would evaluate the system response to each strategy, based on relevant physical, chemical, and biological processes and automatically choose the particular strategy that is optimal in regard to crop productivity and environmental acceptability. The automatic choice of such a strategy is the goal of dynamic optimization. Although dynamic optimization is computationally intensive, the feasibility of using dynamic optimization at the farm level over a sequence of years under different choices of crop rotation, spatial variability, and investment in irrigation systems has been demonstrated by Knapp (1992a, b, c). However, as interacting physical, chemical and biological processes complicate the system, dynamic optimization becomes more impractical computationally. In such cases, the optimization method can be used to screen various options, followed by the evaluation of selected options through detailed process-oriented simulation models already described (Knapp and Wichelns 1990).

Dinar et al. (1993) attempted to provide a framework for policy analysis in regard to lands under irrigated agriculture in arid environments, using dynamic programming models as a quantitative analysis tool. They applied such a model to conditions in evidence on the Westside and used empirically observed input functions for crop yield, salinity, and drainage discharge. The objective function in the model consisted of cost and revenue functions. By evaluating alternate strategies for drainage control, they inferred that direct drainage control policies were slightly more cost-effective than indirect control policies. This research also suggested that dynamic models may not always be necessary for this class of problem because under some scenarios, the system under study may rapidly converge to a steady state, eliminating the requirement that time be an independent variable.

The quantified modeling of the economic and social aspects of agricultural- and environmental resource utilization is a task of enormous complexity. On a local scale, at the level of a farmer, economic analysis focuses on cost-benefit analysis. On a larger scale, as diverse users compete for available water, the allocation of water, especially during periods of drought, becomes a very difficult task. One approach to overcome this difficulty is to let the marketplace decide the value each competing user attaches to its needs. For example, an agriculturist with appropriative water rights may trade water to an urban user. Nevertheless, as amended in California's Constitution in 1928, water must be used for "reasonable and beneficial purposes." As a consequence, economic analysis must consider the relative weights assigned to different segments of society that need water, including agriculture, fish and wildlife, and urban communities. Determination of such weights transcends the gamut of the physical sciences.

The chapters that follow explicitly recognize that physical sciences and policy sciences should come together in a rationally sound manner to meet the challenges of San Joaquin Valley agriculture during the coming decades. As a step in that direction, this work presents our current state of knowledge in regard to physical attributes of agriculture in the western San Joaquin Valley and has attempted to show how such knowledge has a bearing on economics and developing sound policy-related decisions.

3.5.3 Regional Planning Models

A category of management models, distinctly different from the process-oriented and optimization models, is needed for the regional scale of the Westside. Obviously, these models are broad in scope, as they combine resource response with economic objectives. Two important consequences arise from this enlarged scope. First, the number of parameters to be considered becomes very large due to spatial variability of physical properties, spatial variability of agricultural activities, and spatial distribution of economic parameters. Secondly, physical and chemical processes that are defined on the microscale become less and less meaningful and more difficult to scale up as the spatial scale becomes very large. Consequently, large-scale management models must be restricted to fewer parameters, which are accomplished by combining groups of parameters into aggregated model parameters, to simulate in a generalized fashion, the most important hydrologic, geochemical, and agronomic processes and relationships.

These regional-scale models often use empirical relationships derived from smaller-scale, more detailed models, since models at the regional scale are usually difficult to calibrate and impossible to validate. These models are commonly used in planning studies that are concerned with comparisons of the effects of a potential future scenario with the effects of a no-action or base condition. Since the basis of these models is comparison rather than prediction, they play a useful role in basin planning as a means of evaluating alternate strategies. Two examples of planning models among the many models dealing with salinity are the San Joaquin Valley Drainage Program (SJVDP) Westside Agricultural Drainage Economics (WADE) Model and the Hydrosalinity Model (HYSAM) (Hatchett et al. 1989; Aragues et al. 1990).

The WADE Model was used by the SJVDP to make projections of irrigation technology change and drainage production under a series of policy options and constraints on drainage loading to the San Joaquin River. The model used optimization to determine profit maximizing behavior, based on crop revenues and the costs of agricultural production and drainage disposal. Decision variables included crop selection, water supply, irrigation technologies, drainage investment, groundwater pumping, drainage recycling, water transfers between regions, and land use. The model divided the Westside of the San Joaquin Valley into discrete cells of between 6,800 ha (15,000 acres) and 18,000 ha (40,000 acres). Flow between adjacent cells was calibrated against the USGS regional groundwater flow model (Belitz et al. 1991) using a simple Darcy flow assumption and a horizontal conductance term. The WADE Model performed mass balance for flow and salts on the root zone, the shallow semi-confined aquifer, the deep semi-confined aquifer, and the sub-Corcoran confined aquifer.

The HYSAM (Aragues et al. 1990) is a simple mass balance model for salt and water that can be applied to an area of any size on the Westside of the San Joaquin Valley. Although designed for a single annual time step, the model was subsequently adapted to perform sequential annual mass balances and thus simulate salinity transients on the shallow semi-confined aquifer.

3.5.4 Expert Systems

To educate farmers and farm advisors about the effects of improved scheduling and water conservation practices, a graphics-driven expert system known as AGWATER was developed for the California Department of Water Resources by California Polytechnic State University at San Luis Obispo. This computer program prompts the user for information about local climate, soil conditions, cropping practices, irrigation technologies, and how these technologies are managed. The user then selects an irrigation application schedule and, using average California Irrigation Management Information System (CIMIS) data, the water distribution uniformity and efficiency of irrigation for the model plots. In this way, the software user can experiment with a wide range of management and irrigation options, testing new ideas and management strategies before committing to them in the field. The software is described by its developers as an expert system, since it provides the user with recommendations based on the entered field, crop and climatic data and the answers to specific management questions posed by the computer program. For the computer-literate farmer, the advent of computer graphics and shell scripts for generating easily understood user interfaces has greatly enhanced the utility and usability of many simple models, which previously were strictly the domain of irrigation consultants and university researchers.

3.6 Issues and Questions

The regulatory and planning response to the selenium crisis, embodied in the "Rainbow Report" in 1990, ushered in new challenges to irrigated agriculture on the Westside (SJVDP 1990). Optimization of agricultural productivity with minimal consideration of the environmental consequences was no longer a viable option. The Westside's "natural resources" include surface waters, groundwater, wetlands, and fish and wildlife habitat. Sustained utilization of these resources in the broader sense entails grappling with issues of resource quantity and quality on several different spatial and temporal scales. Competition among legitimate but conflicting objectives of resource use necessitates making judgments driven by social, economic, and political considerations, transcending the gamut of science. Problems of agriculture and irrigation can no longer be resolved simply by a construction of another reservoir, storage facility, or a drain system.

Nonetheless, science has to provide to society a carefully documented account of alternatives and consequences, with the understanding that a better-informed society will make better judgments. Having summarized the current status of knowledge of agriculture and drainage on the Westside, the UC Salinity Drainage Task Force proceeded to consider the relevant scientific issues and questions to determine how they might apply to development of in-basin, in-Valley salinity/drainage solutions. The focus was on the physical sciences, including hydrology, geochemistry and biogeochemistry, with emphasis on their application to management and to addressing the biogeochemistry of trace elements, such as selenium.

In California, governmental and social institutions, academic researchers, and industry, at the local, state, and national levels, have increasingly struggled to find a viable balance between short-term and long-term objectives and a balance among competing water resource management visions in the last 30–50 years. Tangible economic benefits, such as annual return on investment, are now resolved on farm and regionally in the context of issues having a longer time scale, factoring in the sustainability of resource infrastructure and impacts on the environment, ecology, and society. These issues span a time scale of decades to centuries. In the physical, environmental, and biological sciences, a corresponding growth in "interdisciplinary research" reflects the recognition by society that the earth is a small planet and all human activities, scientific, technological and social, are interlinked. This recognition underscores the need for long-term planning that addresses the less tangible issues discussed in this section. On the west side of the San Joaquin Valley, the short-term issues typically concern agricultural productivity, while the long-term issues are related to potential changes in resource infrastructure, environmental and ecological impacts, and social values.

3.6.1 Irrigated Agriculture in a Changing Social Climate

During California's brief history since 1850, alliances have been struck at various times between the major competitors for California's developed water supply. After the California Gold Rush, when many of California's eastside rivers and streams were dammed and used to further hydraulic mining, agriculture in the Sacramento-San Joaquin Delta flourished, followed by agricultural development within the Sacramento and San Joaquin Basins. Agriculture was seen as the driving force for urban expansion in California and both went hand-in-hand, culminating in the construction of the Central Valley Project (CVP) and the State Water Project (SWP). Environmental consequences of water development and exploitation became an issue in the 1960s and 1970s, creating a division between municipal and industrial, agricultural and environmental users of water. In the 1980s, the Kesterson crisis and the growing concerns about non-point source pollution, in general, and pesticides, salts and trace elements, in particular, demonized agriculture in the view of many urban coastal communities, eroding the claim that irrigated agriculture has had on the lion's share of the State's developed water. The passage of the Central Valley Project Improvement Act (CVPIA) in 1992 (Tables 1.1, 1.2, 1.3, and 1.4) resulted in the transfer of almost 10 % of agriculture's share of the State's developed water to fulfill environmental demands. Similar shifts in the allocation of water to various beneficial uses occurred with the Bay-Delta Accord and numerous project-specific reallocations of supply were made from urban and/or agricultural uses to environmental resource applications. Many crops cannot compete for water supply at present market rates and hence the future will likely result in changes in cropping patterns and additional reallocations of water supply among agricultural and urban, municipal, and environmental entities.

3.6.2 Farm-Scale Water Use, Drainage, and Productivity

Agricultural productivity depends on activities on individual farms, and it follows that the salinity problem has its source at the individual farm level; therefore, the farm-scale is a rational starting point for a discussion of issues. The most critical issues at the farm scale pertain to optimal management of moisture, aeration and salinity in the root zone. Management of root zone salinity involves dynamic processes of moisture movement and geochemistry in the vadose zone, influenced by the water table below, the atmospheric interface above the land surface, and imported irrigation water. The coupled physical processes that occur within this system are quite complex. Beginning in the 1980s, new instrumentation, automated real-time data acquisition systems (e.g. Childs et al. 1993; Grismer 1992) and computers to aid interpretation (Suarez and Simunek 1993), began to make real-time quantitative analysis available to growers to address complex issues related to management of irrigation and its benefits and impacts.

A key parameter in estimating crop yield is the ET during the growing season. Commensurate with current ability to gather data in the field and to process these data, simple, empirically formulated ET models are being used to estimate the root extraction function and to be incorporated into analysis of moisture-salt dynamics in irrigated soils (Nimah and Hanks 1973; van Genuchten 1987; Ayars and McWhorter 1985; Cardon and Letey 1992a). Improvements in real-time data gathering abilities and interpretation of data are needed in the future to obtain a better estimate of ET and plant uptake functions, perhaps as a function of space and time (growth stage). The plant uptake models, at the level of sophistication available at the time of the selenium crisis in the mid-1980s and the associated development of programs for drainage regulation and management, treated the plant root purely as a physical entity, definable in terms of a prescribed moisture potential or osmotic pressure. However, the plant root is a complex, multifaceted biological entity. It seems likely that future research may incorporate the biophysical properties of the root directly into root zone hydrologic models, thereby going beyond the need to pre-define an ET function.

Plant ET is not the only component of the dynamic root-zone system. The larger system includes the influx and evaporation of water at the land surface and the fluctuating water table below. The two important issues, namely leaching of the root zone and contribution of the water table to plant uptake are interrelated within the larger system. Controlled leaching of root zone salts downward or the controlled supply of water to the root zone upward from the water table are subject to the hydraulic driving forces in the vicinity of the root zone. Although previous workers, e.g. Hoffman (1990), and Grimes and Henderson (1986), have treated these two hydraulic driving forces as separate issues, there exists a need to integrate the analysis of root zone leaching with water table management in a dynamic fashion.

The advent of coupled geochemical and transport models in the earth sciences (e.g. Simunek and Suarez 1993; Liu and Narasimhan 1989) suggests that new quantitative understanding of the hydrogeochemical processes within the vadose zone will be developed over the coming years. Within the vadose zone of an irrigated field, as on the Westside, the nature of the geochemical processes will change from season to season. For example, irrigation waters, saturated with oxygen and containing nitrate derived from fertilizers, will introduce a strong oxidizing environment during the growing season. Changing oxidation states of the pore water will influence not only the precipitation and dissolution of redox-sensitive trace elements, such as selenium and arsenic, but also the sorption properties of oxide minerals and clays. The work of Suarez and Simunek (1993) has already recognized the importance of coupling carbon dioxide production and transport with transient fluid flow variations in the unsaturated zone. There exists a parallel need to couple the highly transient fluid flow in the unsaturated zone with redox-driven reactive chemical processes. The development of such an understanding will involve a combination of carefully controlled geochemical sampling and soil physical measurements in the field, coordinated with the development of new mathematical models capable of combining the transient flow of water with dynamically changing conditions of oxidation and acidity within the soil. It is known that redox-driven chemical transformations in the soil are often mediated by microbial processes. Although the importance of microbial-mediated kinetics in redox-driven chemical reactions is widely recognized, much new knowledge remains to be gathered on how such processes can be described in terms of measurable field parameters and incorporated into computational algorithms.

It is worth noting here that salinity and drainage problems on the Westside have in the past been almost exclusively addressed in terms of total dissolved solids (TDS) and toxic trace elements. To date, contamination of groundwater by agricultural pesticides is an issue that has not been a focal point of research and management. This issue of contamination by pesticides is part of a major, ongoing investigative effort by the U.S. Geological Survey through its NAWQA (National Water Quality Assessment) Program.

Whether one analyzes solely the flow of water or the coupled flow of water and chemicals within the vadose zone on the Westside, the spatial variability of soil properties (hydraulic conductivity, soil moisture capacity, porosity) and forcing functions (infiltration, non-uniform irrigation) render the analysis extremely difficult. Under conditions of spatial variability, it is necessary to use carefully chosen sampling methods, without which field data may lead to misleading interpretations. Hanson and Grattan (1990) provided a discussion of different sampling methods and their relative merits. Once spatially distributed samples are collected, statistical methods should be used to decipher their spatial correlation structures, if any. These correlation structures provide clues about the scale by which the data can be reasonably interpreted under conditions of spatial variability. Guitjens and Hanson (1990) summarized available geostatistical methods for interpreting spatial variability in salinity. Stochastic methods have been employed by Hopmans et al. (1991) to interpret field experiments involving spatial variability in regard to water flow and salinity. For certain types of processes, such as infiltration, it may be possible to account for spatial variability by measurements made at carefully chosen locations (Childs et al. 1993). However, it is not certain that all processes are amenable to simplification to account for spatial variability. Assessing spatial variability of flow and salinity, both in terms of field measurements and in terms of mathematical modeling, will continue to be an important issue in Westside agriculture.

The issue of spatial variability and scale related to chemical transformations is, as yet, a relatively new field (Higashi et al. 2005). Even as chemical thermodynamic models are being developed for redox processes and their kinetic modifications, measurement of redox state in the field is known to be extremely difficult. Frequently, careful field measurements reveal the simultaneous presence of aqueous redox couples in a water sample, which negate the definition of a single redox state for the macroscopic water sample. In systems characterized by such disequilibria, important processes appear to occur at different spatial scales; identity of these processes may be masked by conventional macroscopic averaging processes. Therefore, to handle this type of spatial variability, it may be necessary to formulate new mathematical approaches in which processes occurring at smaller scales are allowed to interact with those occurring at larger scales, which has led to the concept of multiple interacting continua (Pruess and Narasimhan 1985). Several challenges lie ahead in order to develop a clearer understanding of the chemical processes of the vadose zone and to permit the sustained utilization of land and water resources on the Westside without damage to the environment.

Although more sophisticated methods are needed to quantify physical and chemical changes in the root zone and how these changes affect the soil biota and are in turn affected by the soil biota, the ultimate challenge is to translate the knowledge into improved irrigation strategies at the farm level, either for a single irrigation event or for a whole irrigation season. Development of reliable expert systems and decision-making tools to aid farmers and regulators is thus an issue of practical importance. Dynamic optimization models provide a method to help resource allocation to competing users for simple systems. For complex systems, one may need to use detailed process-oriented simulations with trial and error analysis of alternate strategies before arriving at optimal planning or management decisions. Whatever the approach taken, process-oriented models and optimization models need to be compatible in order that the latter may draw upon the most credible input data and the most valid means of characterizing processes. Because each farm is unique in regard to soil, hydrogeologic and geochemical conditions, site-specific models need to be constructed for individual farms, with the desired goal being a computer-literate farmer able to run the model and formulate credible resource allocation decisions on the farm. Work along these lines is already being conducted and will continue to play a useful, practical role in farm-level decision making.

3.6.3 The Link Between Irrigation and the Water Table

The correlation between continued irrigation and rising water tables and salinity on the Westside is well established. Researchers who have carefully studied the performance of subsurface tile drains on the Westside (Deverel and Fio 1991; Grismer and Woodring 1987) have shown that drains can capture water by upward movement from substantial depths. Since the major zone of salt contamination on the Westside lies between about 10 m (30 ft) and about 45 m (150 ft) below the soil surface (Gilliom 1991), flow paths intercepting water at depths greater than 10 m (30 ft) can contain elevated levels of salt and trace elements, such as selenium and boron. The recognition of the importance of upward groundwater flow and its consideration in drainage design and analysis on the Westside are an important first step in placing root zone hydrology in the broader perspective of the regional groundwater system. For long-term management of root zone salinity and the water table on the Westside, it is essential to recognize the relationship between the local scale and regional scale groundwater systems. The movement of water in the root zone and in its vicinity is dictated by two interacting driving forces; one stems from the regional system, driven by recharge on the Coast Ranges, and the other is driven by the potential energy of the applied irrigation water at the surface of the irrigated field. Additionally, large scale pumping of groundwater will also influence water table elevations.

In considering the linkage between irrigation water and the water table (which represents the top of the local shallow groundwater system), it is necessary to recognize that local groundwater flow need not necessarily be horizontal (lateral flow). Indeed, as a consequence of regional groundwater hydraulics, the local groundwater flow field below the water table could at places be more vertical than horizontal. In addition, zones of stagnation will exist within the flow system. The hydrodynamics of the shallow groundwater system may have significant importance with regard to long-term salinity changes, local hydrogeochemistry, and the performance of subsurface drains.

It is known from first principles of regional groundwater motion (Toth 1963) that the disposition of shallow groundwater systems is sensitive to local changes in physiography. Moreover, in the Valley trough and distal margins of the alluvial fans, where discharge areas exist for intermediate and deep flow systems, the potentiometric field driving groundwater motion can be quite complex. Superimposed on these complex patterns are the strong impacts of irrigation and evapotranspiration. In order to analyze these systems, it is necessary to choose a scale that is larger than the farm scale. At this scale, not only will local physiographic features be given due consideration but also the aquifer characteristics and geology of the semi-confined zone will be considered. A need exists to understand groundwater flow systems on a scale that is intermediate between the farm scale and the Valley-wide scale. In order that the link between irrigation and the local groundwater system is properly understood, potentiometric and geochemical data should be gathered from carefully designed piezometer monitoring wells. Ideally, observations and analyses at this scale will provide a link between the local farm-scale processes and regional, district-wide processes.

Closely related to the linkage of irrigation and the groundwater system is the disposition of the subsurface drainage systems. Drainage configuration, depth, and spacing can influence the mixing that occurs of water drawn from the semi-confined aquifer and the deep percolating water above the drain. Inspection of the topology of existing drainage networks can also give clues to the dynamics of flow in the regional aquifer. Many tiled fields are installed using a trial and error approach: new tile lines are installed between existing lines until the requisite drainage yield is obtained and water tables are stabilized. Hence, high drainage densities are indicative of areas of discharge in the regional aquifer. Unfortunately, installation of drains and active pumping of groundwater wells can markedly change a previously stable groundwater regional flow system. Hence, drainage topology may reflect a previous groundwater condition prior to installation of the drains and may or may not be helpful in characterizing current regional aquifer flow. Understanding the interactions between the drain system and regional groundwater flow is thus an important issue.

3.6.4 Monitoring Systems

The time scale at which a specific control volume within the groundwater flow system responds depends on its position within the groundwater aquifer with respect to the ground surface. Shallow groundwater systems within a couple of meters of the land surface may respond relatively rapidly on a scale of days to weeks. Groundwater flow patterns at depths in excess of 6 m (20 ft) may respond on a time scale of weeks to months, while at depths greater than a few tens of meters, one may expect changes occurring over tens of years. At any scale, how the system changes is governed by its capacitance, which is the system's ability to take water into storage in response to increasing external hydraulic pressure. Pioneering work related to land subsidence on the Westside by Poland and Davis (1969) has shown that fine-grained sediments possess much larger capacitance than coarse-grained sediments. The actual change in the nature of the groundwater system in response to irrigation, during a given season or over successive seasons, will be strongly dependent on the distribution of the fine-grained and coarse-grained materials in the groundwater aquifer, as well as changing conditions at the aquifer boundary with the atmosphere, such as rainfall, evapotranspiration, and irrigation.

To approach rationally the issue of long-term groundwater resource sustainability on the Westside, a network of piezometer clusters could provide a record of transient groundwater conditions, including both flow and water quality. The importance of installing and operating piezometer networks and water quality monitoring wells on the scale of a physiographic unit cannot be overemphasized as a tool to guide decisions that could affect long-term groundwater resource sustainability. Work needs to be initiated on the nature, number, and distribution of such monitoring stations and the resources needed to operate them.

The U.S. Geological Survey has collected a wealth of data over nearly four decades on the phenomenon of land subsidence induced by groundwater pumping in the San Joaquin Valley. These data contain much valuable information on the hydraulic properties of the semi-confined aquifer and the confined aquifer and can still be used to help answer questions related to long-term sustainability of the groundwater aquifers and of irrigated agriculture on the Westside if supplemented with more recent information.

3.6.5 The Regional Salt Balance

To achieve long-term resource sustainability on the Westside, the problem of salt accumulation in the regional groundwater system must be reversed or, at the very least, diminished. Salts from Westside agriculture that cannot be disposed of in the San Joaquin River, the San Francisco Bay, and/or the Pacific Ocean must be managed within the Basin. Salts remaining in the Basin may be stored in the root zone, or flushed into the shallow groundwater aquifer, or drawn into the deeper aquifer by high vertical gradients and recirculated to the surface through groundwater pumping.

Salt storage decisions are made at different time scales. On the time scale of a crop irrigation season, management decisions must be made about the disposal of subsurface drainage. On a time scale of decades, decisions may have to be made to retire certain crop lands that are salinized beyond the point of economic recovery, owing to inadequate drainage or because of competing commitments for water deliveries, competing water requirements of wetlands and in-stream flows or to satisfy increasing municipal and industrial requirements. In theory, the productivity of crop lands on the Westside could be maintained indefinitely, if the imported salt load was equal to that exported, assuming an equal and even distribution of salts within the system. In practice, however, this goal may be unattainable, largely because of economic, legal, and institutional constraints and the tremendous heterogeneity of soil and water resources within the system.

Over the next several decades, the region will likely attain some sort of steady state in which a constant quantity of salt is imported into the region each year. For example, Orlob (1991) estimated that by the year 2007, if the then-current trend continued, the net annual importation of salt load would remain approximately steady at about 2×10^6 Mg (metric tons) over the entire Valley. More recent estimates are marginally lower (Schoups et al. 2005). The basic question, then, is how and where this excess salt can be stored or disposed of so that (a) the root zone is maintained at an optimal salinity, (b) the groundwater aquifer is maintained, such that the quality of agricultural pumpage is unaffected, and (c) impacts to other resources are at acceptable levels. Current trends indicate an increasing salt concentration in pumped groundwater in many parts of the Basin.

Problems of rising water tables and salinity buildup are characteristic of the Valley trough. The buildup of salt in the lower parts of the Westside can be divided into two major components: (a) dissolved chemicals transported by regional groundwater flow and (b) salt dissolved in the water imported via the Delta-Mendota Canal and the California Aqueduct. First, the regional groundwater system is largely driven by the topography of the Coast Ranges and of the Westside. The land slopes more or less to the east-northeast (ENE) over the entire area of the Westside. Conforming to this topography, groundwater moves from the recharge areas on the west to the ENE, towards the Valley trough and the San Joaquin River. Mean groundwater accretions along the main stem of the San Joaquin River have been estimated to be approximately 0.06 $m^3\ min^{-1}\ km^{-1}$ or 2 $ft^3\ s^{-1}\ mile^2$ (Kratzer et al. 1987). Flow is greater from the Westside where water tables are marginally higher than on the eastside where water tables are more influenced by high levels of groundwater pumping along the upper reaches of the main stem of the river. Groundwater flow occurs within the Sierran Sand aquifer beneath the Valley trough, from west to east, in response to a groundwater gradient, generated by eastside pumping (Belitz et al. 1991). In contrast, flow within the confined aquifer beneath the Corcoran Clay appears to be in the opposite direction, from east to west, in response to the large volume of groundwater pumping in the upslope areas of the Westside.

Second, despite the fact that water imported from the Delta contains relatively low levels of dissolved salt, the volume of water is so large that the total salt load imported and broadcast on Westside agricultural lands is significant. Orlob (1991) estimated that between 1930 and 1990, the net accretion of salt due to irrigated lands on the Westside amounted to about 75×10^6 Mg (metric tons).

In the early nineteenth century, a major source of salts in the Valley trough would have been transported according to regional groundwater motion. Part of this salt load was concentrated through natural evaporation of water; part of it was stored in the semi-confined groundwater aquifer; and part of it migrated to the San Joaquin River where it mixed with in-stream flows, discharged to the Delta and ultimately to the Pacific Ocean. Following the introduction of irrigation and its rapid adoption in the San Joaquin Valley, salts imported with the irrigation supply have been added to those transported by regional groundwater motion. The pre-irrigation mass balance of salts in the San Joaquin Basin has been perturbed greatly. The ultimate long-range issue of salt-load management in the San Joaquin Basin is how to overcome the excess salt load, given those factors affecting agricultural sustainability that constrain the accumulation of salt in the Basin, and given annual constraints to the assimilative capacity for salt in the San Joaquin River.

Early in the 1960s, the U. S. Bureau of Reclamation (USBR) as well as the State of California planned a master drain (the San Luis Drain) to export all excess salts to the Delta and ultimately to the Pacific Ocean. However, the construction and operation of such a Drain was stopped by the Kesterson crisis of the early 1980s, which led to denying the Westlands Water District (WWD) a drainage outlet to the ocean. The export of salts and trace elements in agricultural drainage from Westside agriculture that has drained historically to the San Joaquin River is restricted to the limits of the River's assimilation capacity, which is currently determined at Vernalis Gaging Station by an EC standard of 0.7 dS m^{-1} during the critical irrigation period and 1.0 dS m^{-1} during the non-irrigation season. These salinity objectives were crafted to protect sensitive agricultural crops in the South Delta. The USBR has been required to make releases from New Melones Dam to ensure that the 30-day running average salinity concentration does not exceed these objectives. In addition, a pipeline to the ocean was investigated by the SJVDP. Although the scheme was technically reasonable, it proved unacceptable for social and political reasons, even before the studies on its cost and possible alignment had been completed.

The plugging up of drains over an area of roughly 2,150 ha (5,300 acres) in the WWD to eliminate selenium-contaminated drainage from Kesterson Reservoir has led to a pressing need to find in-basin solutions for the disposal of agricultural drainage. As a result, re-use of agricultural drainage and the blending of agricultural drainage to irrigate salt-tolerant crops has increased. Drainage return flows arising after such re-use are being used to grow salt-tolerant trees. Some return flows from these areas are discharged to evaporation ponds where these are available. The UC Salinity Drainage Task Force and its drainage management partners were initially faced with a moratorium on the construction of new evaporation ponds on the Westside due to problems of bird safety. Over the past 20 years, carefully monitored drainage experiments have demonstrated that evaporation ponds that are well-mitigated may be managed with reduced impacts on water birds and shore birds.

The fact that the zone of high salinity and trace element contamination has migrated to between 20 and 50 m (60 and 150 ft) of the land surface (Gilliom 1991) is also an indication that the groundwater system is storing salts generated by long-term irrigation. Letey and Oster (1993) analyzed recent data pertaining to rates of water table rise on the Westside and concluded that allowing the water table to remain close to the root zone supplies a portion of plant ET (thereby reducing irrigation needs) and also encourages marginally increased downward flow into the deeper semi-confined aquifer. Based on estimates of vertical hydraulic conductivity of Corcoran clay, Letey and Oster (1993) reasoned that increased vertical flows could help to maintain a steady, shallow water table over long periods of time. They also suggested that it is necessary to evaluate the relative merits of storing salts in the groundwater system over extensive areas, as opposed to disposing large quantities of salt in small areas through evaporation ponds.

Although Letey and Oster (1993) have helped to focus attention on the importance of the groundwater system with respect to salt management, the transient response of the semi-confined zone to irrigation is, in fact, more complex. The semi-confined zone on the Westside varies in thickness from 125 to 300 m (400–900 ft) and comprises both fine-grained and coarse-grained sediments derived from the Coast Range. In the lower part of the Valley, coarse-grained sediments derived from the Sierra Nevada Range also exist. Available geological knowledge suggests that fine-grained sediments (clays, silts and sandy clays) occur intermixed with coarse-grained sediments, such as gravel, sand and silty or clayey sands and gravels. The heterogeneous system appears to consist of a complex system of layers, lenses, and stringers of different parent materials. Because of their shallow, unconsolidated nature, the fine-grained sediments tend to be highly compressible and hence characterized by high specific storage (hydraulic capacitance).

As a consequence of the sloping topography of the Westside and the complex system of layers, lenses, and stringers, the flow pattern within the semi-confined zone is likely to be characterized by lateral sub-horizontal flow paths in the coarse-grained units and vertical flow paths within fine-grained units. Moreover, because of the presence of compressible fine-grained sediments, the manner in which the effects of a fluctuating water table propagate downwards to the Corcoran Clay is complex in both space and time. It is likely that the effects of a fluctuating water table would be damped by the compressible formations within the semi-confined zone. An understanding of how the effects of managing the water table propagate down through the semi-confined zone, causing changes in groundwater storage in the fine-grained sediments, is of fundamental importance.

An understanding of how the migration of the salinity front moves downward through the heterogeneous semi-confined system over long periods of time and how the rapidity of water quality changes at depth in the semi-confined system are of critical importance. Continuous monitoring of potentiometric heads in piezometer clusters installed at appropriate locations in the watershed is crucial for this purpose. Installation and maintenance of such observation stations distributed over the Westside must be considered an intrinsic part of the long-term management strategy for the Westside of the San Joaquin Valley.

3.6.6 Conjunctive Use of Water

Conjunctive use of groundwater will greatly aid in managing the hydrologic regime in the vicinity of the root zone. Groundwater pumping could help lower the water table and alleviate a need for drainage. Apart from the fact that such a lowering of the water table will reduce the availability of plant ET from the water table, the relationship between water table fluctuation and water pumping in a large heterogeneous system with highly compressible aquitard materials is complex. If the water table is to be managed over large areas, the number of wells required, their distribution, depth, and the schedule of pumping would need to be determined. For this purpose, it would be necessary to characterize hydraulically the semi-confined zone to a much greater detail than has been attempted to date. Furthermore, it is important to recognize that significant pumping of groundwater from above the Corcoran Clay can have serious consequences. First is the potential for land subsidence arising from the slow compaction of fine-grained sediments. Second, the drop in potential in the semi-confined zone will profoundly influence leakage through the Corcoran Clay (Belitz and Phillips 1993, 1995).

3.6.7 Valley-Wide Water Management

At the regional scale, the management of irrigation water deliveries, drain systems, and the chemical quality of the agricultural return flows on an inter-district water scale is an issue of relevance to institutions, such as the USBR, the California State Water Resources Control Board, the California Department of Water Resources (DWR), the California Regional Water Quality Control Board, and the State Legislature. These agencies require the use of workable tools, not only for operational purposes but also for long-term planning purposes.

The USBR and DWR are the responsible federal and state agencies, respectively, concerned with water resource planning in the San Joaquin Valley. The most significant planning decisions, made annually, that affect the agricultural productivity of the land resources and the environmental quality of riverine and groundwater resources are the water allocation to federal and state water contractors. These decisions are made with the help of mathematical models that perform monthly accounting calculations matching water demands with the available developed water supply in the CVP and SWP reservoirs, primarily located in the Sacramento Valley. During times of drought, water deliveries to water districts are cut back, resulting in reduced irrigation applications to large areas and increased groundwater pumping, particularly on the west side of the San Joaquin Valley, which relies most heavily on imported Delta water.

3.6.8 Mathematical Models

The interrelationships that exist between the natural groundwater flow system and irrigation activities on the land surface have been broadly captured in mathematical simulation models of the western San Joaquin Valley or of the San Joaquin Basin to date, although the detail required to turn these simulation models into decision support tools is still inadequate. To make rational decisions that might help to achieve long-term sustainability of surface water and groundwater resources on the Westside, the operation of this large, active, and variable system has to be continuously fine-tuned. Such fine-tuning must rely on two major efforts: monitoring and modeling.

The first principle is that complex and dynamic natural systems evolve continuously. Some parameters of the system can only be estimated with adequate data in the time domain, while other parameters may change with time. Therefore, a network of monitoring stations to continuously observe the hydrogeological and hydrogeochemical attributes of the system is an essential, integral component of long-term planning for the Westside. Second, with regard to interpretation and forecasting, it is essential that adequate tools are available to analyze the monitored data. At the present time, computer-based numerical models offer the best interpretative tools for the purpose. Therefore, assembling a set of relevant computational tools is an issue of major importance. As we have seen, problems of interest vary in scale from that of a single farm to the Valley as a whole. In the time domain, the scale of interest may vary from a single irrigation event to several decades. On a small scale, process-oriented models for water movement, heat transport, and reactive chemistry are important. On the largest scale, the operation of distribution networks plays a more important role than process-specific details. Yet, small-scale and large-scale models must be compatible with each other. One way to achieve such an end is to think of a hierarchy of interrelated models, linking issues on all spatial and temporal scales.

3.7 Looking Forward

As we focus on the sustainability of irrigated agriculture on the Westside at a time of global competition in agriculture, the need to maintain a healthy environment in California and to support a rapidly increasing population render the following issues relevant to irrigated agriculture on the Westside:

3.7.1 Short-Term Production-Oriented Issues

- Evaluation of crop ET needs as a function of changing conditions of soil moisture and salinity.
- Design of irrigation scheduling and application methodology based on changing soil physical conditions, crop ET needs, and disposition of the local water table.
- Incorporation of information on spatial variability of soil properties and topographic changes into techniques for evaluating crop ET needs as well as the design of irrigation schedules and water application methods at the farm level.

- Development of optimization models on a time scale of seasons to integrate irrigation practices, cropping patterns, and economics.
- Integrated operation of drainage-networks (Grasslands District) to discharge effluents into the San Joaquin River at times of high assimilative capacity to comply with water quality standards prescribed at downstream locations. The impact of these operations on the long-term changes in selenium accumulation in the San Francisco Bay.
- Development of real-time monitoring instruments and data transmission equipment at the farm level as well as regional levels to provide data for estimating crop ET needs, irrigation scheduling, and management of drainage networks.
- Improved methods of water reuse, crop selection, and agro-forestry in portions of the Westlands Water District (WWD) where no natural outlet exists for disposing drainage effluents and where trace element contamination is a more serious problem than salinity.
- Conjunctive use of groundwater and surface water to manage water table elevations and reduce drainage volumes and contaminant loads.
- Extension of the useful life of pumping wells by controlling groundwater degradation.
- Assessment of the potential for real-time monitoring and deployment of improved information transfer technologies to reduce drainage volumes and contaminant loads.
- Development of cost-effective techniques to control erosion and sediment transport from the intermittent streams of the Coast Ranges, which periodically deposit large volumes of selenium-laden sediments on agricultural lands situated on alluvial fans.

3.7.2 Long-Term Sustainability Issues

- Assuming that the San Joaquin River is the only outlet available for exporting saline effluents, the rate at which net salt accumulation will accrue on the Westside even under the most efficient water use scenarios.
- The extent and depth to which the groundwater system has been contaminated by past agricultural practices.
- The nature and the time scale of coupling between the irrigation system, the regional groundwater system, and regional pumpage. The time rate at which irrigation-mobilized contaminants propagate through the groundwater system.
- The relationship between pumpage from aquifers above and below the Corcoran Clay and the potential for induced contamination of the deep aquifers across the Corcoran Clay through irrigation wells.
- Determination of the hydrological and geochemical linkages among irrigation, drain-networks, and the regional groundwater system on a scale of topographic or physiographic units.

- Delineation of areas where land degradation occurs irretrievably and the implications for land retirement, contract renewal, and long-term land management.
- Developing monitoring programs to monitor systematically the long-term response of the land and the groundwater system to irrigated agriculture.
- In the context of justifiable environmental concerns and growing urban needs for water, the development of methods for determining the "net public good" emanating from agriculture on the Westside and exploration of the economics of conversion of irrigated land to other beneficial uses.
- Resolution of water policy issues: If, according to the amendment of California's Constitution of 1928, all water in California must be used for reasonable, beneficial purposes, can users of water trade their quotas for money? Or, should users forfeit the allocation of water, if it is not used for the purpose for which it was intended?

This long list of issues is addressed at a variety of spatial and temporal scales in later chapters, which discuss research and field-testing to accomplish the following goals:

- Define the origins of the trace element problem (focus on selenium (Se) but address other potential ecotoxic elements in the soils) and the various transport, transformation, and bioaccumulation pathways for ecotoxicity;
- Develop a better understanding of the biogeochemistry of Se and other trace elements in the environment, leading to strategies for remediation;
- Develop and test in-basin management alternatives and their potential roles in an integrated approach to managing irrigation, drainage, and the impacts of drainage on wildlife; and
- Develop a basis and tools for evaluating the various remediation and management options to determine their relative costs and benefits to help optimize management.

Research and development since the Kesterson crisis have been the basis for a number of on-farm and regional efforts to address the issues and to broaden understanding of the temporal and spatial scales of the needed solutions.

Acknowledgments The author acknowledges decades of productive collaborations of his colleague, N. T. Narasimhan. This work started as an educational exercise on the part of a hydrogeologist (T.N. Narasimhan) and an earth scientist and water resources engineer (N.W.T. Quinn) to synthesize our understanding of the important technical issues pertaining to irrigated agriculture on the Westside of the San Joaquin Valley. Because of the breadth of the technical questions involved, much new material beyond our areas of expertise was digested. We are indebted to the following for thoughtful, critical reviews of the first draft of this manuscript: James E. Ayars, John Fio, Wilford R. Gardner, Mark E. Grismer, Edgar A. Imhoff, Keith C. Knapp, John Letey, Theresa Presser, J. van Schilfgaarde, Donald L. Suarez, K.K. Tanji, H.J. Vaux, Jr. and Wesley Wallender. We are thankful to Manucher Alemi, Kenneth Belitz, Sally M. Benson, and David Zilberman for constructive criticisms of the revised manuscript. This work was partly supported by the Salinity and Drainage Task Force of the University of California and partly by the Regional Research Funds of the Agricultural Extension Service.

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Chapter 4
Scales and Scaling as a Framework for Synthesizing Irrigated Agroecosystem Research on the Westside San Joaquin Valley

Wesley W. Wallender, Jan W. Hopmans, and Mark E. Grismer

4.1 Introduction

The inherent complexities of the hydrology and geochemistry in irrigated agriculture and the need to integrate theory with experimental conditions require innovative and interdisciplinary research efforts. A major research challenge in irrigated agriculture is to understand the relevant properties and processes of agroecosystems across a wide range of spatial and temporal scales (Narasimhan and Quinn 1996; Wallender and Grismer 2002). The problems of irrigated agriculture and drainage occur at the plot, field, farm, watershed, and regional scales and occur at different time scales. The concepts of process – observation or measurement, model, and domain or system-scales as well as scaling these processes from small to large scale – have provided a framework to synthesize and connect the broad range of research on sustainability of irrigated agroecosystems over the last 25 years. The challenge of scaling is summarized by Grismer (2007):

> Irrigation hydrology is constrained to analysis of irrigated ecosystems in which water storage, applications, or drainage volumes are artificially controlled in the landscape and the spatial domain of processes varies from micrometers to tens of kilometers while the temporal domain spans from seconds to centuries. The continuum science of irrigation hydrology includes the surface, subsurface (unsaturated and groundwater systems), atmospheric, and plant subsystems. How do we scale up highly nonlinear physical, chemical, and biological processes understood at natural scales to macro- and mega-scales at which we measure and manage irrigated agroecosystems? How do we measure, characterize, and include natural heterogeneity in scaling nonlinear processes?

All of the problems associated with irrigated agriculture and drainage are driven by water and its interactions with soil. Water is thus the logical integrating factor for studying agricultural, environmental and economic sustainability of irrigated

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_4,

agroecosystems because it links processes (disciplines) across space and time scales. The challenge is to use water as the integrating link in the analysis while fully recognizing that water quantity and quality only partially affect the sustainability of the agroecosystem in the San Joaquin Valley. Narasimhan and Quinn (1996) comprehensively cast their efforts in the context of scale and occasionally mention scaling. The thrust in this chapter is how scaling was used in the various research projects conducted by the University of California Salinity Drainage Task Force in the San Joaquin Valley in the past 25 years. Describing and illustrating scaling issues and related approaches in irrigation/drainage hydrology and their application in California's Central Valley will help irrigation-drainage engineering/science professionals better address scaling problems, management plans, and public policies in formulating designs affecting irrigated ecosystems.

4.2 Continuum, Scales, and Scaling

Water connects biological, chemical, physical, economic, and other processes over a range of scales within an agricultural production system. Water influences and is influenced by transport and transformation of air, solutes, soil and biota vital for life, spanning the continuum from groundwater through the unsaturated zone (including the root zone) to the atmosphere. Individual hydrologic processes are spatial-dependent (from plot to region) and time-dependent (day to decade) and therefore, because agroecosystem sustainability is related to biophysical processes, water-related sustainability is process-scale-dependent.

Scales refer to the spatial and temporal dimensions of active processes that occur in the irrigated landscape, which influence the timing and scale of data collected and observations recorded. The natural process scale is defined as a characteristic dimension in time and/or space associated with a process, while scaling refers to concepts/paradigms linking processes occurring at different space or time scales. Fundamentally, many transport processes, including the transport of water, salinity, and trace elements, occur and are understood at space and time scales associated with laboratory and bench top measurements that are then applied through some type of averaging at the larger field and watershed dimensions. Differing perspectives on spatial and temporal scales also influence the definition of problems and the formulation and implementation of solutions.

In analyzing and managing water-dependent processes in irrigation, scaling involves linking phenomena observed at various spatial and temporal scales. The intended outcome is an understanding of the phenomena from a system perspective, an understanding of how the processes are linked and interact. The processes important to irrigated agriculture can be addressed in terms of four typical spatial scales and four temporal scales (Table 4.1).

Microscale is the smallest scale at which a system can be considered a continuum. Microscale functions (such as pore water processes) are generally addressed at the laboratory level. Below this scale, materials are viewed as discrete particles

Table 4.1 Spatial and temporal scales relevant to several important water processes in irrigated hydrology[a]

Temporal scale	Spatial scale: Micro (laboratory) (10^{-5}–10^{-2} m)	Macro (greenhouse) (10^{-2}–1 m)	Mega (plot – field) (1–10^4 m)	System (watershed/district) (10^4–10^8 m)
Seconds	Water properties Pore-water processes Ion-exchange	Porous media flow Pipe flow	Crack flows	NA
Hour-days	Root uptake	Infiltration Transpiration Precipitation	Irrigations Surface flow Leaching ET Field drainage	Precipitation Canal deliveries GW recharge flows
Years	Some soil chemical reactions	Plant growth	Leaching Soil changes	Watershed processes Atmospheric processes GW flows
Centuries	Geochemical processes	Pedogenesis processes	Watershed changes	GW contamination

[a]ET and GW denote evapotranspiration and groundwater, respectively

and properties, such as water content and flow velocity, and are not continuous functions. In contrast, macroscale functions (such as plant transpiration) occur and are often measured at laboratory or greenhouse scale and values from this scale are averaged for purposes of addressing them at larger spatial scale or longer temporal scale. Megascale refers to functions at the field and watershed scales; small spatial variations such as the patchy distribution of various soil types are often ignored and the system is usually described using averages. At the system scale, such as the watershed or district, functions such as precipitation or net volume of drainage provide insight into the functioning of the entire system of interest. Gas, liquid and solid phases are commonly considered as overlapping (volume-averaged) continua.

To understand, interpret, and model various processes, observations are often made at measurement scales (dimensions) smaller than the actual process scale. If the measurement scale differs from the natural scale, the resulting quantification of a process may not precisely reflect the process of interest, but rather some average thereof. Thus, a change in measurement scale may affect what is observed, may change the process studied, and finally, may alter the mathematical description of the process, particularly if the scale-dependent processes are non-linear. A critical challenge for researcher is to identify the measurement scale that best matches the observed and modeled biophysical process(es). The model scales (e.g. time-and-space-steps used in a numerical groundwater model) are often increased beyond the measurement scale to reduce data and computation requirements. However, the model may generate errors, if the process scale is approached or exceeded, particularly for heterogeneous systems. The model scale for the independent variable(s) in such processes or functions may be different from that of the dependent variable(s).

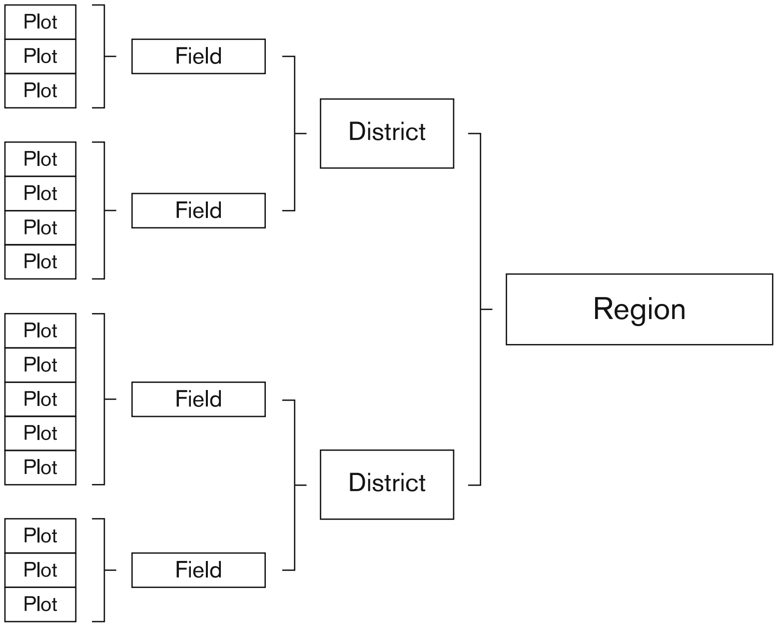

Fig. 4.1 System spatial scales from plot to region

On the other hand, it is often not possible to measure processes directly at the model scales deployed, so an understanding or correction of modeled processes must be noted or otherwise accounted for.

The system scale is commonly, but not always, greater than the scale of all the processes considered (Fig. 4.1). The spatial and temporal boundary of the region is assigned according to where and when information about the system and its boundary is known or approximated and according to where and when information about the system and boundary is sought. In general, the system spatial scale may change with time. If the watershed defines the system scale, then the initial and boundary conditions of its water-affecting systems must be defined.

The difficulty in scaling up and down is gathering a set of concepts that enables process information or models developed at one particular scale to be used in making predictions at another scale (aggregating or upscaling). There are many issues that make this linking of scales challenging and sometimes impossible due to the non-linearity and heterogeneity of the landscape. This is especially the case for irrigated agriculture, where soils are heterogeneous, and nonlinear hydrologic processes control drainage and soil salinity, both critical elements for effective irrigation water management. Harvey (1997) and Bugmann (1997) identify six causes of scale and scaling problems that apply in hydrologic systems:

- Spatial heterogeneity may occur in surface/subsurface processes,
- Non-linearity in system responses may occur at different scales,
- Processes may require threshold scales to occur,

- Dominant processes of concern may change with different scales,
- Emerging processes may develop, resulting from mutual interaction of small-scale processes, and
- Disturbance regimes (e.g. dams, canals etc.) may be superimposed on natural systems.

When considering such scaling issues, one particularly difficult challenge is upscaling dynamic physical, chemical, and biological processes understood at microscales to macroscales that can be managed and measured, and then upscaling to the megascales of agroecosystems at which local, county, state, and national policies may apply. During upscaling, the introduction of simplifying assumptions and the averaging of natural heterogeneity may compromise interpretation of upscaled models. For example, a field may have substantial heterogeneity in soils, with patches of clays and loams differentially distributed on the surface and in the soils below the surface. Irrigation applied to a field with such patchy soils may infiltrate at varying rates and have varying effects on crop growth and on drainage. Thus, the second challenge is measuring and characterizing the natural heterogeneity. Measurement scale and system-scale can be upscaled as well. This is commonly required when the demand for information and computation is excessive and when lack of information does not compromise analysis.

Upscaling systems involves identifying the spatial and temporal boundaries of the larger region, according to where and when information about the larger system and its boundary is known or approximated and according to where and when information about the larger system and boundary is sought. That is, by upscaling, domain boundaries increase, and the definition of boundary values involves using smaller-scale observations. Dis-aggregation is defined as downscaling from the larger to a smaller spatial/temporal scale and may be necessary if the natural scale of the process is a vital factor in the assessment of interest. In upscaling and downscaling, nonlinearity and heterogeneity within the domain cause information loss and/or boundary value domain errors.

Study of each process and the upscaling and downscaling of each process are generally organized by “discipline”; yet, understanding and predicting sustainability often emerge from multiple processes within and among disciplines. To connect processes across disciplines and to transfer information among processes at common or disparate scales, upscaling and downscaling are generally required (Fig. 4.2). Because information is compromised during upscaling and downscaling, errors propagate into the evaluation and prediction of sustainability. Moreover, because observation and model scales differ among disciplines, issues arise when integrating disciplinary results, as required when solving interdisciplinary problems, such as agricultural sustainability. Additional error may be introduced in linking discipline-to-discipline data. Consequently, linking across disciplines in interdisciplinary research is critical and typically involves scaling, in order to transfer information between spatial/temporal scales.

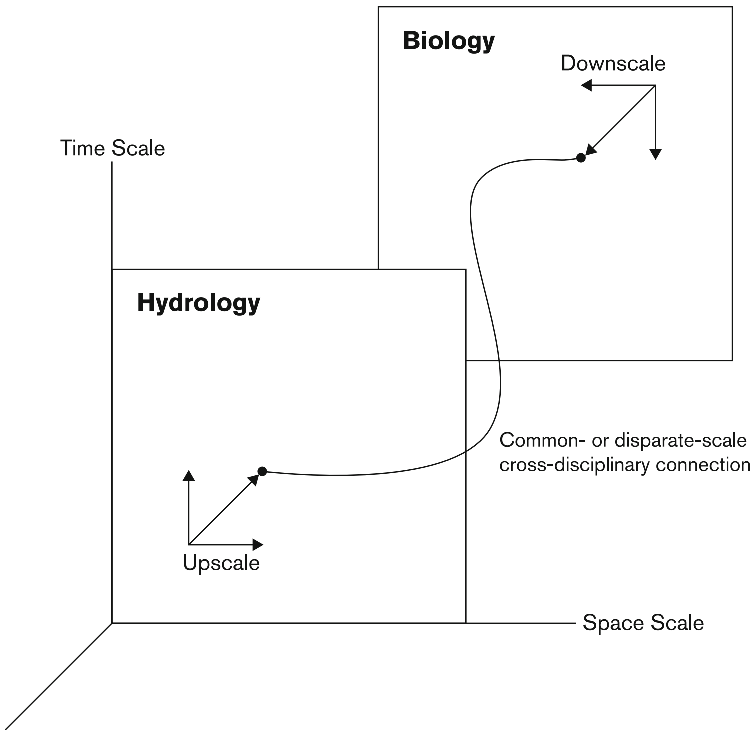

Fig. 4.2 Scales and scaling within and among disciplines

4.3 From Microscale to the Westside of the San Joaquin Valley

In the research that has occurred since the Kesterson crisis and in the application of that body of research to drainage and selenium management, there were numerous upscaling and interdisciplinary linkage challenges. For example, at the "small pore scale," a microscale located in the porous media of soils and groundwater systems, a multiphase collection of air, water, and solids occurs (Table 4.1). The Navier Stokes equations describe fluid flow within the pores only, and constitutive equations relating stress and strain tensors characterize the deformation of the solids in the porous matrix. Because the geometry of the interfaces as well as that of the boundary conditions is impossibly complex, flow equations are spatially smoothed across the fluids and solids to develop equations that are applicable everywhere in the bulk volume (Whitaker 1999).

In the following, we present examples and discussion that illustrate the relevance of scales across the space-time continuum of irrigated agriculture in the San Joaquin Valley and several examples of upscaling through the entire continuum of scales. The first example is a case study of scaling, focusing on soil hydraulic functions across the range of scales from the pore scale to the regional scale. Subsequent examples focus on specific spatial scales, but stress the bridging of disciplines required, acknowledging the enormity of the breadth and depth of sciences needed for a comprehensive knowledge of the principles, impacts, and policy understanding of irrigated agriculture.

4.3.1 Scaling Soil Hydraulic Functions: Microscale to System Scale

4.3.1.1 Theory and Method Considerations

Understanding the movement of water through soil is essential to understanding and addressing salinity and drainage issues in the San Joaquin Valley. Irrespective of scale, transient isothermal unsaturated water flow in non-swelling soils is described by the Richards equation, which provides the soil water matric potential (h_m), water content (θ) and water flux density as a function of time and space, using one-, two-, or three-dimensional flow models:

$$C(h_m)\frac{\partial h_m}{\partial t} = \nabla \cdot [\mathbf{K}(h_m)\nabla(h_m - z)] \tag{4.1}$$

Where K is the unsaturated hydraulic conductivity tensor (L T^{-1}), z denotes the gravitational head (L) to be included for the vertical flow component only, $C(h_m)$ is the soil water capacity and represents the slope of the soil water retention curve.

The equation captures the two critical variables that determine water flow through soils: the capacity of the soil to retain water (soil water retention) and the rate of water movement through a porous medium (hydraulic conductivity). When combined, these two variables are referred to as soil hydraulic functions. Soils vary in terms of both functions, and there is enormous variability in soils at many scales. Alluvial soils of the San Joaquin Valley are highly variable and patchy (heterogeneous) in their distribution.

Both the soil water retention capacity and the unsaturated hydraulic conductivity equations are highly nonlinear, with both h_m and K varying many orders of magnitude over the water content range of significant water flow. Gravels and sands have very low water retention and high hydraulic conductivity; clays have high soil water retention and low hydraulic conductivity. Because the functions are not linear, applying Eq. 4.1 across spatial scales is inherently problematic. Specifically, the averaging of processes determined from discrete small-scale samples may not describe the true soil behavior involving larger spatial structures and the resulting error may be large. Moreover, the dominant physical flow processes may vary between spatial scales. Because their measurement in the field is time-consuming, relatively few sites may be sampled and the measured hydraulic data are usually insufficient, less than required statistically to characterize fully the heterogeneity of the soils. As a result, data assimilation techniques, such as linear regression analysis, pedotransfer functions, and neural networks have been developed to derive soil hydraulic functions from other, easier-to-obtain soil properties. Considering that soil hydraulic measurements are typically conducted for small measurement volumes and that the natural variability of soils is enormous, the main question asked, is how small-scale measurements can provide information about large-scale flow and transport behavior. These issues associated with spatial scaling of soil hydrological processes were presented in Hopmans et al. (2002a).

Soil water retention and unsaturated hydraulic conductivity relationships must be known *a priori* to solve for unsaturated flow and salt transport in the vadose zone. They are determined by mutual relationships between soil water matric potential (h_m), volumetric water content (θ) and hydraulic conductivity (K). Functional soil water retention and unsaturated hydraulic conductivity models are based on size distribution, geometry, and connectivity of pores. Measuring these microscale soil characteristics directly is time-consuming and difficult; their prediction based on easier-to-obtain soil data (i.e. soil texture) is uncertain; and relationships are highly nonlinear, with both h_m and K varying many orders of magnitude over the water content range of significant water flow.

Soil hydraulic functions at larger spatial scales are obtained primarily from relatively small measurement scales. For example, prediction of soil-water dynamics at the field-scale is routinely derived from the measurement of soil hydraulic properties from laboratory cores, collected from a limited number of sampling sites across large areas, resulting in sampling sites spaced at considerable distances apart from one another. Typically, the measurement scale for soil hydraulic characterization is on the order of 10 cm, with a sample spacing of 100 m or larger. Routine measurements of soil hydraulic properties are usually conducted in the laboratory (Dane and Hopmans 2002), using small-size soil cores, collected *in-situ* from a few representative or many random locations at the system scale, or collected in the field on small field plots at the meter-scale using either direct or inverse methods (Hopmans et al. 2002a). Soil parameters obtained from cm-scale measurements (laboratory scale) are included in numerical models with a grid or element size ten times as large or larger, with the numerical results extrapolated to field-scale conditions. Because of the high nonlinearity of soil hydraulic functions, their application across spatial scales is inherently problematic.

Upscaling requires integration and aggregation of spatial information into larger spatial units, e.g., as in the estimation of an effective field soil water retention curve or conductivity curve from small-scale laboratory core measurements. One may differentiate the microscopic pore scale, the local scale, and the regional scale, such as agricultural fields and watersheds. The macroscopic Darcy equation is considered to be valid for the laboratory scale, with a typical size in the range of centimeters to meters. It is also this scale for which the Darcy equation can be derived from the volume averaging of the Stokes equations at the microscopic scale level. The regional scale typically applies to agricultural fields and watersheds for which the relevant soil hydrological properties become increasingly nonstationary. Yet, it is at these larger spatial scales that solutions are increasingly sought.

Laboratory-measured soil hydraulic properties are appropriate input for laboratory-scale column studies and simulation models. When applied to larger spatial scales, two alternatives may be considered. First, the laboratory-scale soil hydraulic properties may be spatially distributed across the larger spatial scale of interest, assuming that integrated flow behavior can be determined from the averaging of many individual, soil "column-like" flow processes, with the spatial variability estimated using the scaling approach. Second, laboratory-measured soil hydraulic functions can be used as initial estimates, improved by inverse modeling to be discussed below, and conditioned by scale-appropriate large-scale boundary conditions and flow measurements.

The scaling approach has been used extensively to characterize soil hydraulic spatial variability and to develop a standard methodology to assess the variability of soil hydraulic functions and their parameters. The single objective of scaling is to coalesce a set of functional relationships into a single curve using scaling factors that describe the set as a whole (system scale). The procedure consists of using scaling factors to relate the hydraulic properties at a given location to the mean properties at an arbitrary reference point. This physically-based scaling concept provides for the simultaneous scaling of the soil water retention and unsaturated hydraulic conductivity functions (Clausnitzer et al. 1992), leading to scaled-mean soil hydraulic functions for the system scale under consideration, to serve as effective soil hydraulic functions.

Inverse Modeling (IM) offers another approach for extracting the intrinsic linear dynamics that govern a complex system directly from observations of the system. Practically, IM provides accurate approximations of dynamics that become more exact as the system becomes more random or chaotic. The inverse method offers a powerful procedure to estimate flow properties across spatial and temporal scales in systems with highly diverse or chaotic characteristics, such as fields with variable soil and irrigation practices. As numerical models have become increasingly sophisticated and powerful, IM methods are applicable to laboratory and field data and are no longer limited by the physical dimensions of the soil domain, or the type of imposed boundary conditions. Methods of IM might be especially appropriate for estimating regional-scale effective soil hydraulic parameters, using boundary condition measurements. For example, Eching et al. (1994) estimated field-representative hydraulic functions using IM with field drainage flow rate serving as the lower boundary condition for the Richards flow equation applied at the field-scale. The application of IM to estimate soil hydraulic functions for laboratory soil cores has been reviewed extensively by Hopmans et al. (2002b). Although the IM method may not capture the unique characteristics of a given plot or field, the application of IM methods, in general, to estimate soil hydraulic functions across spatial scales is very promising, yielding effective hydraulic properties that pertain to the scale of interest.

4.3.1.2 Scaling from Pore-to-Laboratory Scale

Much of the difficulty in predicting soil hydraulic functions lies in their non-uniqueness, the lack of a firm theoretical foundation of the hydraulic relationships, and the possible control of retention and conductivity functions on the flow dynamics of the measurement, in addition to pore geometry characteristics. Wildenschild et al. (2005) studied this 'dynamic effect' using x-ray computed microtomography (CMT) with a spatial resolution of 6–20 μm. This nondestructive and noninvasive imaging technique allows water flow and salt transport to be visualized at the pore scale. Of particular concern is the adhesive force between water and soil (matrix potential); strong matrix potentials bind water to the soil and require energy (pressure) to induce movement. Flow experiments were conducted in 6-mm diameter soil columns that were packed with a sandy soil material with a mean particle size of 0.17 mm. Multi-step and single-step outflow experiments were conducted (Eching and Hopmans 1993; Hopmans et al. 2002b) to investigate the influence of drainage flow rate on soil water distribution in the drainage pore space. The CMT analysis confirmed that large soil water matric potential gradients can cause water to be entrapped in the draining soil core, thereby increasing soil water retention and reducing soil hydraulic conductivity, irrespective of the soil water matric potential in the soil core and possibly affected by the soil core diameter, relative to the mean particle size. These experiments demonstrate the difficulty in measuring key hydraulic functions in the laboratory for application in the field and expose how error in laboratory-scale experiments may be introduced to the scaling process.

4.3.1.3 Scaling from Laboratory-to-Field Scale

Most unsaturated water flow and transport models have been developed for applications at the field and larger spatial scales. Eching et al. (1994) compared laboratory-scale soil hydraulic functions with field-scale effective hydraulic properties for a 32-ha furrow-irrigated field on the Westside of the San Joaquin Valley. In this specific application, *in-situ* field-measured drainage curves using neutron probe soil moisture measurements were entered into a one-dimensional unsaturated water flow model to optimize soil hydraulic functions by IM, comparing the model-predicted outcome with the field-measured depth distribution of soil water content at 44 locations. Boundary conditions were determined from irrigation and ET estimates during 125 days of the growing season. Using the scaling approach of Clausnitzer et al. (1992) and Vogel et al. (1991), field-averaged and spatially-distributed laboratory-measured soil hydraulic functions were obtained from undisturbed soil cores, collected from the same 44 field locations and compared to model predictions. The resulting effective soil hydraulic properties of soil water retention were surprisingly close to the field measurements. Yet, the laboratory-measured saturated hydraulic conductivity values were approximately one order of magnitude larger than those estimated from the field data. The study concluded that effective hydraulic properties were able to describe the average transient soil water behavior for the heterogeneous field soil systems.

4.3.1.4 Scaling from Field-to-Regional Scale

Though feasible, a prohibitively large number of sampling sites are needed to characterize the spatial scales of the vadose zone larger than the field scale. An alternative approach is to estimate effective values for the hydraulic parameters using neural network analysis, whereby soil hydraulic parameters are estimated from easy-to-measure soil properties that are readily available from soil maps. Such an approach was used in the regional study of Schoups et al. (2005a, b), using the three-dimensional hydrosalinity model MOD-HMS to determine spatially distributed hydrologic flows and salt transport for a 1,400 km^2 irrigated area in the western San Joaquin Valley. MOD-HMS is a MODFLOW-based distributed watershed model that simulates subsurface flow and transport in an integrated manner, including deep recharge and regional groundwater flow. The adapted modeling approach is based on the coupling of a soil chemistry module with MOD-HMS, to yield an integrated approach for simulating three-dimensional variably-saturated subsurface flow and reactive salt transport. In addition to estimated soil hydraulic functions, input data included spatially distributed crop types and weekly irrigation and rainfall amounts for each field of the region.

The 1,400 km^2 modeled area (domain) was divided (discretized) into a regular finite difference grid of 2,960 square cells corresponding to a typical field size of 64 ha (0.25 km^2) or 805 m (0.5 mi on each side). In the vertical direction, the model domain extended from the land surface to the top of the Corcoran clay, using 17 layers of increasing thickness from the surface downwards. The total number of active model grid cells was 36,040. Hydrologic flows and salt transport were simulated for a 57-year period, from 1940 to 1997, using annual average boundary conditions and grid-cell-specific soil parameters. The salinity module included reactions, such as cation exchange, precipitation, and gypsum and calcite dissolution. Using historical crop acreage and water delivery records for each water district, crops and irrigation amounts were randomly distributed, leading to the annual assignment of a single crop to each grid cell. Other required boundary conditions were needed to quantify vertical (across the Corcoran clay and into deep groundwater) and lateral (towards the San Joaquin River) water flow, salt fluxes, and exchange between the simulated domain and its surroundings, so that an annual salt balance could be estimated. Spatially distributed water flow, salinity reaction, and transport parameters were obtained from soil survey data and 242 well logs. The model was used to reconstruct historical changes in salt storage by irrigated agriculture during the past 60 years. The model analysis indicates that patterns in soil and groundwater salinity were caused by spatial variations in soil hydrology, the change from local groundwater to snowmelt water as the main irrigation water supply, and occasional droughts. Gypsum dissolution was a critical component of the regional salt balance. Although results show that the total salt input and output were about equal for the past 20 years, the model also predicts salinization of the deeper aquifers, thereby questioning the sustainability of irrigated agriculture.

4.3.2 Scaling in the Analysis of Subsidence

Scaling in the analysis of subsidence is an example of upscaling from the nanoscale to the regional scale. The western San Joaquin Valley region experienced significant land subsidence from 1940 to 1970, caused by the heavy use of groundwater resources (Poland et al. 1975). With additional surface water supplies in the late 1960s, pumping rates declined dramatically, significantly decreasing land subsidence. However, during subsequent drought periods, reduction in surface water supplies has resulted in increased pumping and subsidence rates comparable to those prior to 1970 (Larson et al. 2001). Though land subsidence is not a current problem in the region, due to high water tables from inadequate drainage, groundwater pumping has been proposed as a means to alleviate the high water table, salinity accumulation problem. Such pumping increases the threat of renewed subsidence, especially when pumping to meet increased water demands for irrigation is included, associated either with drought or global warming effects on the availability of water resources in the San Joaquin Valley. Thus, groundwater pumping-induced land subsidence must be balanced against shallow water table management and drainage reduction; appropriate modeling and analyses are needed for the management of groundwater resources on a long- and short term basis.

The physical phenomena governing the relationship between water extraction from confined and unconfined aquifers and land subsidence were analyzed using a microscopic linear momentum balance to create the needed models. The analysis begins with consideration of individual soil or sand grains comprising the porous media. For a single grain at rest in a saturated medium, the temporal rate of change of momentum is zero; that is, there is no net force acting on the particle. In general, forces acting on the grain are body forces due to gravity and surface forces due to (a) pressure exerted by fluid surrounding the grain (shear stress is zero for static fluid) and (b) contact forces due to adjacent mineral grains. With constant gravity forces, any changes in fluid pressure produce a corresponding change in surface forces due to contact forces. The mineral grains are compressible and thus a change in contact forces will affect the grain-to-grain contact area. An increase in contact forces will mean an increase in contact area, which in turn implies a decrease in forces exerted due to water. The combined effect of these factors increases the vertical component of the force vector between the soil particles, which may then cause the soil particles to move, resulting cumulatively in land subsidence. Thus, inter-particle forces in fluid-saturated porous media are one of the main factors influencing compressibility, deformation, and strength properties.

Larger-scale land subsidence is likely to induce changes in inter-particle forces at the microscopic level, which, in turn, also imply changes in contact stresses and contact areas. The inter-particle forces, together with contact stresses and contact areas, are likely to affect the porosity and flow paths of the porous media, thereby directly affecting the storage and hydraulic properties of the aquifer. Apart from the groundwater flow equations, Terzaghi's principle of effective stress and the Kozeny-Carman equation for the saturated hydraulic conductivity of a porous media are the main tools to model the process of aquifer compaction and relate the changes in porosity to the changes in hydraulic conductivity (Singh and Wallender 2008). However, these models of effective stress and saturated hydraulic conductivity do not account for the physical processes of force transmission at the microscopic level; thus, the effects of changes in inter-particle forces, contact stresses, and contact areas due to land subsidence are not reflected in the evaluation of storage and hydraulic properties of the aquifer. In short, the existing models used to predict water flow through porous media such as soil do not account for microscale phenomena and may thus introduce error into land subsidence modeling.

Through upscaling analysis of the physical processes of changes in inter-particle forces, contact stress, and contact area due to land subsidence at the grain level and their effects on the storage and hydraulic properties of an aquifer, Singh and Wallender (2007) found the spatially averaged total solid phase stress in the z-direction for a saturated porous media under hydrostatic conditions to be:

$$\langle \sigma_z \rangle = \langle p \rangle - \langle p \rangle \beta_z + \langle \langle f_{ns} \rangle \rangle_z (1 + \mu \tan \varsigma) \beta_z \tag{4.2}$$

Where $\langle \sigma_z \rangle$ represents the spatially averaged total stress in the solid phase in the z-direction, $\langle p \rangle$ is the average pore pressure, β_z denotes the ratio of projected soil-to-soil contact area to total area of the horizontal plane, $\langle \langle f_{ns} \rangle \rangle_z$ is the spatially averaged contact stress value defined on a horizontal plane due to normal contact stress in the vertical direction z, μ is the friction coefficient from Coulomb's friction law $f_{ss} = \mu f_{ns}$ where f_{ss} and f_{ns} are the magnitudes of the tangent stress vector and the normal stress vector, respectively, at the contact plane between solid particles, and the variable contact angle formed by the contact plane can be replaced by a representative angle ς.

The effects of solid matrix deformation related to the changes in particle-to-particle contact stress enabled derivation of a generalized Kozeny-Carman equation to predict saturated hydraulic conductivity:

$$K = \tau n \left(\frac{\rho_w g}{c \mu_w} \right) \frac{n^2}{(1-n)^2 (1-\chi)^2 S^2 G_s^2 \rho_w^2} \tag{4.3}$$

where τ is tortuosity, n is porosity, ρ_w is the mass density of water, g is the gravitational constant, μ_w is the dynamic viscosity of water, c is a parameter resulting from the conversion of radius of flow capillary to the hydraulic radius, χ is the fraction of the surface area of the soil particles ($0 \leq \chi < 1$) not in contact with the flowing water, S and G_s are the mass specific surface area and the specific weight of solids, respectively.

The connection between the two previous efforts was through χ.

4.3.3 Scaling of Diffusion Through Soil

A second example of upscaling, from pore to centimeter scale, is the dispersion tensor, critical to simulation of salt transport on the Westside of the San Joaquin Valley. Salts and contaminants in groundwater tend to spread and create a diffuse plume rather than moving in a single direction at a constant concentration. Dispersion perpendicular to the aquifer flow direction (downward), known as transverse dispersion, plays an important role in the remediation of contaminants because it helps to dilute their concentration and to mix them with reactive compounds and microbes in the surrounding groundwater.

Movement of water and the minerals dissolved in water is both vertical and horizontal. It is important to be able to calculate the rate of dispersion through the various soils in the irrigated area and down slope. Dispersion through the soil is a function of the size of the pores between the grains of soil, the length of the pores, the friction, the lenticular bedding, layering of the soils, and fractures in the soils. Dispersion is thus a relatively complex process to model. In upscaling, it is important to recognize the potential for error in estimating the rate of longitudinal or horizontal flow.

The method of volume averaging (Whitaker 1999) from fluid in the pores to the bulk volume is applied to ordered and disordered spatially periodic porous media in two dimensions in order to compute the components of the dispersion tensor for low Peclet numbers (a ratio of advection to diffusion) ranging from 0.1 to 100 (Buyuktas and Wallender 2002). They found that the longitudinal dispersion coefficient decreases with an increase in disorder, while the transverse dispersion coefficient increases; that is, water disperses in the horizontal plane less rapidly in complex patchy soils and disperses downward faster. The location of discs within the unit cell considered for analysis influences the longitudinal dispersion coefficient significantly, compared to the transverse dispersion coefficient. For laminar flows, the dispersion coefficient is independent of the particle's Reynolds number (at low rates of flow, the Reynolds number is dominated by viscosity considerations) and the predicted functional dependence of dispersion on the Peclet number agrees with experimental data. The predicted longitudinal dispersion coefficient in disordered porous media is smaller than that of the experimental data. However, the predicted transverse dispersion coefficient agrees with the experimental data. This underprediction of longitudinal dispersion would cause under prediction of vertical salt migration from shallow groundwater to the deep semiconfined aquifer through the Corcoran Clay layer and to the confined aquifer on the west side of the San Joaquin Valley.

4.3.4 *Scaling from the Plot to the Regional Scale*

4.3.4.1 Spatial Heterogeneity in Soils

When numerical models are used for estimating the water flow and dissolved mineral (solute) transport at field scale, modelers are confronted with heterogeneity, particularly the issue of spatial variability of soil hydraulic properties within the field and also the issue of selecting model element size which is smaller than the process length scale. Scaling up the spatial variability from the plot scale to the field scale involves estimating the effective soil hydraulic parameters, which includes the net effect of the heterogeneous/patchy soil conditions on water movement through the soil. Buyuktas and Wallender (2004) estimated the field scale effective soil hydraulic properties that faithfully reproduced measured water and solute transport to tile drains using a three-dimensional deterministic model. This was done by direct averaging of field soil characteristics to interpret the field as a single equivalent soil. Such effective (net averaged) soil hydraulic properties can be used for heterogeneous soils in numerical simulation models to simplify calculations.

4.3.4.2 Water Flow and Transport

Another scaling issue is how to determine the effectiveness of water flow and transport of dissolved salts to subsurface drains, which provides important data for the size and spacing of the drainage system elements. Tarboton and Wallender (2000) developed an algorithm to size model elements to be less than the process length scale in a finite-element model for the purpose of calculating flows to drains. By comparing the numerically simulated drain flow rates and head distributions with analytic values, a nested configuration was found to be appropriate for an effective drain radius of 0.01 m, and a square configuration was suitable for an effective drain radius of 0.05 m. Using an analytic solution, a method was developed to determine a drain's distance of influence as a function of its effective radius and the geometry of its flow domain. The distance of influence was found to be independent of material type. An appropriate spacing of lateral drains was selected for the numerical simulation of multiple drains by increasing the grid mesh spacing well outside the drain radius of influence. The position of the water table and the drain flow rate with time were used to evaluate the between-drain grid spacing for transient variably saturated flow. Grid Peclet and Courant numbers, together with the shape of the solute advance front, were used to evaluate the suitability of the selected single drain configuration and lateral drain spacing for solute transport. The resulting finite-element grid configuration for single and multiple drains ensure a stable, efficient numerical solution and has applicability to numerical modeling of multiple subsurface drains. In short, knowing the drain size, spacing, and grid configuration allows models to upscale from the plot to the field or farm level.

4.3.5 Irrigation Practices and Management

Beyond the field scale, regional scale hydrologic modeling that is capable of assessing the impacts of irrigation practices and management strategies on water resources requires the estimation of spatially heterogeneous hydrologic properties. Again, the patchy distribution of different types of soils with different hydrologic characteristics is an upscaling issue. To characterize and investigate the effects of spatially variable hydraulic properties on optimized irrigation and drainage efficiencies at the regional scale in the San Joaquin Valley, Gates and Grismer (1989) used stochastic techniques combined with Monte-Carlo modeling of vadose zone and ground water flows associated with irrigation and drainage of cotton. In contrast, Tarboton et al. (1995) showed how data on soil texture from lithologic and geophysical logs of wells at the plot scale in the western San Joaquin Valley can be used as a basis for estimating regional spatial soil texture distribution, using kriging statistical techniques, which assume that soil characteristics at nearby points are related, but soil characteristics at distant points are statistically random. Surfaces obtained using three-dimensional indicator kriging were compared with those obtained using an inverse distance-weighting technique. Greatest differences between the methods occurred where data with disparate values were clustered. Expected values of kriging, which are less extreme than those from distance-weighting methods, are considered to be more representative of average values over three-dimensional spatial blocks because they include the effect of local variability via the nugget and use vertically correlated data. Anisotropic hydraulic conductivities (conductivity related to directional “grain” of the soils) are estimated from expected values of the kriged heterogeneous textural distribution to obtain improved estimates of regional hydrologic properties that also provide valuable information regarding uncertainty of the estimates.

In modeling, it is important to be able to estimate the uncertainty or variance of predicted values for yields, irrigation efficiencies, or other planning type parameters in developing sustainable irrigation/drainage practices in the San Joaquin Valley. Gates and Grismer (1989), Gates et al. (1989) developed a stochastic vadose-zone, groundwater optimization model that characterized water and salinity transport in the unsaturated (vadose) zone and the saturated groundwater zone at the field (32 ha) and regional (20 km^2) scales to determine the optimal irrigation and drainage efficiencies and associated variances and confidence levels resulting in the greatest economic return from cotton production. In this model, irrigation efficiency was defined as the ratio of crop evapotranspiration (ET) to applied water and drainage efficiency was defined as the fraction of excess irrigation deep percolation captured by the subsurface drainage system. Regional subsurface lateral flows were included. The model employed probability-density functions (random fields) describing the spatial variability of irrigation applications, hydraulic conductivity, upward flow, and seepage to drainage systems at the field and regional scales within a Monte-Carlo simulation approach to address the effects of variability on drainage and irrigation efficiencies.

The impacts of rootzone salinity, water table depth, irrigation and drainage efficiencies, and system costs on optimal economic returns during a 20-year simulation period were included. Time steps were variable during the irrigation season (days to weeks) and weeks to a month during the off-season. In the 20 km^2 region considered, contour maps of expected water table depths, soil salinity, relative crop yields and their variances were generated.

The results for the 20-year simulation period indicated that the systems largely stabilized after about 12 years and that the optimal irrigation and drainage efficiencies were 78 and 91 %, respectively, with regional expected annual return of $162/ha. The model application demonstrated the physical parameters having the greatest effect on expected regional net benefits, including irrigation application rates, spatial variability of hydraulic conductivity, and salt dissolution rates within the root zone. The optimal efficiencies suggested that well-managed surface irrigation systems may be appropriate and that some excess deep percolation should be allowed, such that water is available as upward flow for crop water demands and deeper aquifer recharge. Perhaps more importantly, the stochastic modeling approach provided variances and levels of confidence with respect to planning parameters of importance, including the distribution and confidence levels associated with annual economic returns. Incorporating uncertainty due to variable conditions at the field and regional scales allowed managers to understand the range of probable outcomes for a given irrigation management strategy. In short, this approach enabled system responses to management to be interpreted with the concepts of stability and risk.

As described in many instances in this book, scaling from the farm level to the regional level involves addressing the spatial and temporal perspectives of growers, water managers, and other stakeholders. An interesting comparison is the relatively strong and immediate public and regulatory concern generated during the Kesterson crisis in the early 1980s. Solutions were perceived to be critical and the objectives of the solutions proposed were characterized by (a) a sense of immediacy and (b) a desire for a quick and final solution at the single site. In contrast, the perspective at the Tulare Lake Drainage District, where similar Se toxicity was found in evaporation basins, was characterized by a lower sense of immediacy and a willingness to accept something less than zero impact. The factors affecting this difference in temporal perspective and outcome perspective on the scale of the proposed and acceptable solutions affected the problem definition and the formulation and implementation of solutions. In short, scaling considerations have scientific and technological implications and socioeconomic implications. The concept of scaling thus provides a fundamental framework for discussing a wide range of issues.

4.4 Linking Models at Various Scales

In addition to modeling flow to drains at the field and farm level, modeling the evaporation (discharge) of groundwater due to a shallow water table can be an important component of field and regional scale water balances (e.g. Bali et al. 2001a). In irrigated regions where soil moisture varies on short time scales, this type of modeling is most accurately accomplished using variably saturated modeling codes. However, the computational demands of these models limit their application to field-scale problems and upscaling is required. The MODFLOW groundwater modeling code is applicable to regional scale problems and it has an evapotranspiration package that can be used to estimate this form of discharge; however, the use of fixed (time-invariant) parameters in this model results in evaporation rates that are a function of water table depth only. Young et al. (2007) calibrated and validated the previously developed MOD-HMS model code using lysimeter data. In San Joaquin Valley research, the model was used to illustrate that bare soil evaporation rates depend on water table depth and soil moisture conditions. Finally, Young et al. (2007) have presented a piece-wise linear upscaling approach for estimating the time varying parameters for the MODFLOW evapotranspiration package using a 1-D variably-saturated MOD-HMS model.

Growers in this region face serious agricultural, environmental and economic sustainability problems. In upscaling, it is therefore appropriate to consider a linked hydrologic and economic model of irrigated agriculture on the Westside of the San Joaquin Valley. Because the rate of salt attenuation in the groundwater system is low, the groundwater system is compromised. Highly saline, shallow water tables containing naturally occurring ions such as selenium and boron threaten agricultural productivity and are potentially environmentally damaging. In the San Joaquin Valley, high soil salinity reduces crop yields and may render land useless for agricultural production. The presence of highly saline, shallow water tables creates a need for the disposal of drainage water, which negatively impacts receiving waters. Restrictions imposed on drainage water disposal can directly lead to increased soil salinity. The economic viability of agriculture in the region is further threatened by reductions in surface water supply legislated in the Central Valley Project Improvement Act (CVPIA) and in other recent regulatory actions (see Table 1.1, 1.2, 1.3, and 1.4 in Chap. 1), which may lead to further overdraft of limited fresh groundwater supplies.

4.4.1 Regional Coupled Models

At a regional scale, a layered, integrated model has been constructed for a 900 km^2 region in the western San Joaquin Valley, consisting of a hydrologic model linked to an agricultural production model (Wallender et al. 2002). The hydrologic model is a distributed 3-D variably-saturated flow and transport model with full reactive salt chemistry capabilities (Schoups et al. 2005a). The agricultural production model simulates agricultural production decisions at the water district level. It is assumed that growers maximize profits subject to the pertinent resource and environmental constraints. Given the initial conditions regarding surface water allocation, and soil, surface water, and groundwater salinity, the agricultural production model simulates agricultural production on an annual basis and produces spatially distributed information on the following: cropping patterns, water applications, groundwater pumping, irrigation efficiencies, and crop yields. The output from the agricultural production model is used subsequently by the hydrologic model to simulate the impacts of these management decisions on the natural system. The agricultural production model, in turn, is updated annually by the hydrologic model to account for changes in soil salinity, water table depth, and groundwater salinity. Outputs of the integrated modeling system include district-level farm profits, crop yields, and spatially and temporally distributed values of soil salinity, groundwater salinity, water table depths, and drain volumes and loads.

An important consideration in coupling the economic and hydrologic models is their difference in scale, both in space and time. First, the agricultural production model makes predictions at the district scale across all of its salinity zones; whereas, the coupled hydrologic model uses grid cells at a spatial scale of 2.59 km^2 (1 mi^2). Since there are several hydrologic grid cells within each regional soil salinity zone, upscaling and downscaling operations are conducted to transfer information between the agricultural production model and the coupled hydrologic model. Downscaling is accomplished by randomly assigning crops to hydrologic grid cells within a soil salinity zone, such that the crop acres for that zone, as predicted by the agricultural production model, are preserved. Future versions of the model will contain algorithms that assign crop locations that are most consistent with observed cropping rotations but also aggregate to the regional crop acreage totals. Upscaling from the hydrologic grid cells to the agricultural production zone, on the other hand, is readily achieved by averaging grid cell soil salinities over each zone. Second, the agricultural production model predicts annual irrigation amounts for each crop in each zone. These annual values are distributed over the months for input into the hydrologic model by assuming constant irrigation efficiency throughout the year, i.e. monthly-applied water is computed from monthly crop water demand.

The coupled model was used to simulate the economic and hydrologic responses to various changes in water supply and the management variables. For a 50 % reduction in surface water supply, results of simulations with the integrated modeling system indicated a significant increase in fallowing and a shift from low- to high-value crops. Increasing irrigation efficiencies by investing in new technology is also a possible response, but it was shown to be more likely under nonsaline conditions. The reduction in surface water supply also resulted in a declining water table, and reduced discharge of subsurface drainage water and salts.

This decade-long effort to develop an integrated, scale-dependent analysis seeks to define the sustainability of irrigated agroecosystems in terms of soil, the deep vadose zone, groundwater and surface water quantity and quality; agronomic and ecosystem productivity quality; and finally, economic viability. By upscaling the system to a significant fraction of the San Joaquin Valley, while limiting the upscaling of modeling, the farm-scale effects of a region-scale system policy are possible to estimate and project.

4.4.2 Field and Regional Water Use Efficiency Models

Taking a somewhat different approach, employing measured ET_o (reference evapotranspiration, a generalized measure that allows for regional-level comparisons of evapotranspiration rates) and crop yields across several counties and regions in the southwestern USA, Grismer and others investigated the field and regional Yield/$ET_{c.}$ ratios, or water-use efficiencies of forage hay and cotton crops in comparison to actual field values ($ET_{c.}$ is the specific crop ET rate). The goal of this work was to assess not only the ratios and water-use efficiencies encountered in actual crop production systems, but also to evaluate possibilities of water savings and net water values for a particular crop. The scale of this effort was broader than the other modeling efforts described, and demonstrates how multi-system analyses may provide data for regional policy making.

The water use efficiency model simulations were applied to cotton (*Gossypium hirsutum*) lint, alfalfa (*Medicago sativav*), and Sudan grass (*Sorghum bicolor* subsp. *drummondii*) hay yields, yield-consumptive use ratios (Yield/$ET_{c.}$) and hay prices across a range of rainfall and evapotranspiration conditions in the western states to determine crop water value, or benefit. The analysis included a determination of long-term mean values and variability of yield, Yield/$ET_{c.}$ ratios and associated irrigation water values. These were compared with published hay water-use-efficiencies, production and water costs. Alfalfa hay Yield/$ET_{c.}$ ratios decrease with increasing $ET_{c.}$, though their variability increases with increasing ET_c. The greatest Yield/$ET_{c.}$ ratios (16–17 kg ha^{-1} mm^{-1}) and irrigation water values (\$2,800–\$3,000 ha^{-1} m^{-1}), with relatively moderate variability, are associated with an irrigation water requirement of approximately 800 mm, reflecting a combination of relatively high hay values, $ET_{c.}$ and beneficial rain. Similarly, as with dry matter production, cotton lint yields in interior valley regions of California were weakly correlated with $ET_{c.}$ and averaged 1.33 Mg ha^{-1} (Upland) and 1.08 Mg ha^{-1} (Pima). Cotton lint yields (LY) in desert regions of Arizona and California were not correlated with ET_c. The greatest LY/$ET_{c.}$ ratios (1.9–2.1 kg ha^{-1} mm^{-1}) were in the San Joaquin Valley and were similar to those from water use efficiency studies and resulted in gross irrigation water values (\$3,400–\$3,800 ha^{-1} m^{-1}), with relatively moderate variability at a net irrigation water requirement of approximately 720 mm. While this irrigation water value was twice that of water delivery prices below the California delta and is comparable with average municipal water costs of \$4,200 ha^{-1} m^{-1} for large western cities, the average was nearly three times less. However, the cotton lint irrigation water value was two to three times greater than that obtained for alfalfa and Sudan grass hay crops in all regions.

These studies (reviewed by Grismer 2007) suggested that crop production in the high ET desert regions do not generate the greatest return on water investment in terms of water value, compared to the return from more moderate ET conditions found in cooler areas inland or in some cases along the California coast. Conceptually, Yield/$ET_{c.}$ should not be a function of ET requirement of crop ($ET_{c.}$), and if the desert area (high ET) data are separated, no relationship between Yield/$ET_{c.}$ and $ET_{c.}$ occurred.

Actual Yield/$ET_{c.}$ ratios approached the expected water use efficiencies from greenhouse and lysimeter studies only in the San Joaquin Valley (or along the coast and in Los Angeles where total planted areas are relatively small).

This type of system-scale modeling can inform policy decisions. Based on the model analysis, there appears to be an opportunity to allocate less water to the other areas without loss in yields. The volume of saved water can be determined and compared to anticipated reductions in reservoir releases for agriculture. Perhaps hay and cotton lint production in desert environments may not be tenable, and the water may have greater value in other applications. On the other hand, is it possible to increase hay or lint yields in the desert areas to levels comparable to that found inland? For example, in the Imperial Valley of Southern California, intense summer heat results in relatively low hay yields but high water use, so it has been suggested that summer irrigations be reduced to maintain the hay stand but not achieve significant production. The system-scaling approach is able to address the issue of how much improvement can be obtained by this suggested practice and whether other farm water management techniques exist that will enable greater water use efficiencies to be achieved.

In a related field analysis of alfalfa and Sudan grass hay production on heavy clay soils in the Imperial Valley, Bali et al. (2001b) and Grismer and Bali (2001) found that the reduced-runoff surface irrigation method (a simplified volume-balance model approach to determining irrigation cut-off time or distance developed by Grismer and Tod (1994)) resulted in greater hay Yield/$ET_{c.}$ ratios. In practice, the method requires measurement of a presumably nearly constant onflow rate and a single measurement of surface water advance rate down the field. During studies spanning 3 years, the average alfalfa Yield/$ET_{c.}$ ratio was increased from an estimated Valley average of 8.9–15.2 kg ha^{-1} mm^{-1}. This latter value is comparable to that obtained in high production regions of the southern San Joaquin Valley. Correcting project hay yields for an estimated 30 % reduction associated with an average soil salinity of 6 dS m^{-1} (Maas and Hoffman 1977) suggests that the reduced-runoff irrigation method resulted in a Yield/$ET_{c.}$ ratio of nearly 21 kg ha^{-1} mm^{-1}, a value similar to the maximum WUE expected for alfalfa hay. Similarly, a Yield/$ET_{c.}$ ratio of 15.5 kg ha^{-1} mm^{-1} was obtained for Sudan grass hay production, approximately 15 % less than expected WUE as a result of an estimated 15 % salinity-stress induced loss. Improved Yield/$ET_{c.}$ ratios were obtained in part from limited use of shallow groundwater by the stressed alfalfa crop during its first year of production.

Results from the reduced-runoff irrigation trials, as well as those from the drip and furrow irrigation trials under high soil-salinity conditions in the San Joaquin Valley (Grismer 2001a, b, c), suggest that greater attention should be given to anticipated salinity effects on hay and cotton crop coefficients and subsequent estimations of applied water depths. Overall, allowing for potential depressed yields as a result of salinity stress as well as pragmatic considerations of crop production, average water allocations (neglecting rainfall) sufficient to achieve target water use efficiencies of perhaps 17, 1.7 and 2.1 kg ha^{-1} mm^{-1} for hay, pima and upland cotton lint production, respectively, and ~18 kg ha^{-1} mm^{-1} for forage hay production can be set as target values and compared to Yield/$ET_{c.}$ to determine potential water savings at the farm level per county or area.

The water use efficiency models and field studies provide information of potential use in addressing large-scale issues related to water supply and allocation to meet urban, agricultural, and environmental objectives. The greatest water values and least possible water savings occur in the southern San Joaquin Valley where there is a combination of relatively high ET and minimal rainfall. Nevertheless, improving Yield/$ET_{c.}$ ratios for alfalfa hay and cotton lint production in the southern San Joaquin Valley, partially a service area of Friant Dam, may result in a water savings of 2.4 times the recently mandated water releases for in-stream fisheries. Similarly, possible water savings in the Imperial Valley from hay and cotton production represent nearly 19 % of its entire Colorado River allocation. On the other hand, the combined annual water savings potential from the Imperial and southern San Joaquin Valleys for hay and cotton production total a little more than the evaporation losses at Lake Powell. This research is a starting point for assessing water use/savings at the field scale for hay and cotton production and should be extended to other crops. Additional work may also be required, considering water savings at the district scale associated with water distribution systems.

4.5 Research Opportunities

A research challenge is to continue decreasing the process, model, and measurement scales while increasing the system scale. The need is for both low-cost sensors and methods for field data collection and stronger computers for analyzing, predicting, and animating the behavior of these complex, irrigated agro-ecosystems from small to large system scales. Small-scale sensors will aid and motivate environmental scientists and engineers to study processes at the scales used by chemists, physicists, and biologists.

With better field and analytical tools, research and application opportunities are abundant. They include scaling up fundamental processes in time and space to represent more accurately the irrigated agricultural production system as part of the broader landscape, developing indicators to identify system function (and malfunction) at all scales, and managing production systems, at appropriate time and space scales, from input (e.g., water and nutrient application) and output (e.g., productivity and environmental impact) perspectives. Incorporating error analysis and factors that account for risk and uncertainty in decision making in conjunction with scaling is an accompanying research opportunity. These better tools will enable engineers and scientists to predict the probability of sustaining irrigated agro-ecosystems in California's San Joaquin Valley over the full range of scales from the perspectives of agriculture, the environment, and the economy.

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Chapter 5
Biogeochemistry of Trace Elements: Reactions in Soils

Christopher Amrhein and Harvey E. Doner

5.1 Introduction

Following the discovery in 1983 of selenium (Se) toxicosis in waterfowl that were feeding in agricultural drainage water at the Kesterson Wildlife Refuge, irrigation project managers have become aware of the need to evaluate soils and drainage waters for Se, as well as for other potentially toxic trace elements (Ohlendorf et al. 1986). Soils and sediments are the source of nearly all the trace elements found in water and biological systems. Now, more than a quarter century later, agricultural practices in the San Joaquin Valley have changed to adapt to soils, sediments, and waters containing elevated concentrations of toxic trace elements. Thus, it becomes imperative that we understand the reactions of Se and other trace elements in soils in order to evaluate their environmental hazards. Their chemical properties and reactions in soils are important to understand because they affect mobility, bioavailability, and toxicity. Predictions of their short- and long-term reactions in soils can only be valid, if they are based on sound scientific principles. Prudent land management decisions depend on our understanding of the reactions of trace elements in soils.

Prior to the establishment of the UC Salinity Drainage Task Force to study the problem of Se toxicity in the San Joaquin Valley (SJV), only limited scientific information was available about Se in soils. Geering et al. (1968), for example, reported the solubility and redox criteria for the possible forms of Se in soils. Cary and Allaway (1969) made an important contribution to our

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_5, © Springer Science+Business Media Dordrecht 2014

understanding of the stability of different forms of Se applied to low-Se soils. There are numerous early reports in the scientific literature about the soil chemistry of Se, but most were developed in regions of the world where there were Se deficiencies or toxicities, due to highly elevated concentrations in soils. There were early warnings that Se might be of concern in the soils on the Westside of the SJV (Lakin and Byers 1941) but these were not seriously considered in the planning process for irrigation because Se concentrations in the soils were not high enough to cause toxicity to grazing animals. Changes in management practices resulted in the leaching of Se from irrigated soils and its accumulation at toxic levels in drainage ponds at the Kesterson Reservoir. Understanding Se chemistry in soils under these conditions became the focus of numerous studies. Also, more significance was given to other potentially toxic trace elements in soils, given the surprise of Se in the SJV.

When considering chemical reactions in soils we are normally concerned with applying basic chemistry to processes at the solid-water interface. This means we look at chemical processes on a very small spatial scale and try to extrapolate these to the landscape. Time scale is another very important consideration in chemical reactions. Does a reaction occur within a fraction of a second or over extended periods of time, such as months, years, or centuries? As we will show in this section, time scale is very important when considering whether or not a trace element is toxic. This means we must understand the various forms and species of the trace element in soils.

The biogeochemistry of any element in soils involves its distribution between the total elemental content in the soil and the amount of the element available for intake by an organism or mobile through a soil. Figure 5.1 is a schematic diagram illustrating the distribution of trace elements among their various forms.

The dashed-line box encloses the solid, soluble, and gaseous forms of the trace element, indicating those components that constitute the total trace element concentration in the soil. Knowing the total concentration of an element in a soil may provide very little information about its potential bioavailability. Historically, the total concentration was most often reported with no regard to biologically active concentrations. As shown in the total content box, we have three different phases in soils that provide a source of elements: (a) a solid phase composed of soil minerals and organic matter, (b) an aqueous (liquid) phase consisting of water and its dissolved constituents, and (c) a gaseous phase. With many elements, it is frequently the solid phase that contributes the most to the total elemental content, while the liquid phase contributes the most to the mobile and bioavailable fraction. This is where the time scale for reactions becomes very important. The arrows illustrating dissolution/desorption and precipitation/adsorption between solid and soluble phases can take place on time scales from milliseconds to geologic time, depending on the element and environmental conditions. Whether an element is in a solid phase or solution also controls whether it is available for biological uptake and mobile through the soil. Understanding the soil conditions controlling transformations back and forth from

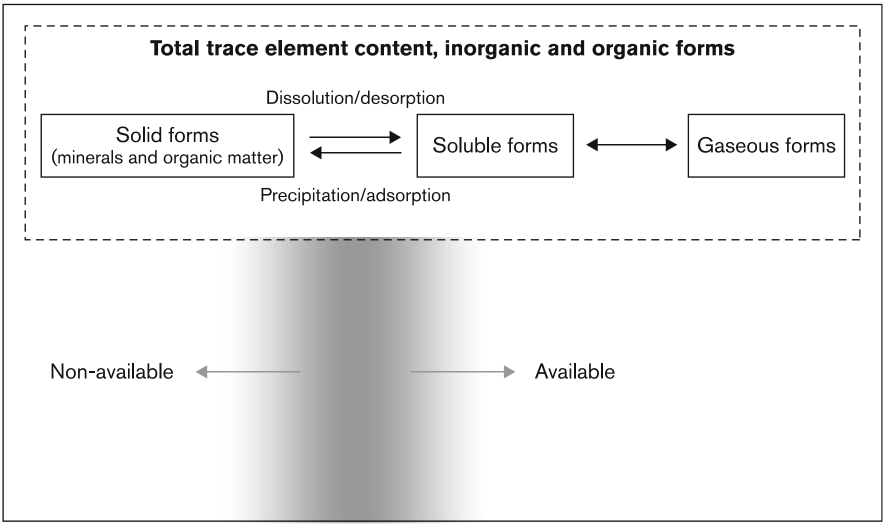

Fig. 5.1 The distribution of trace elements in solid, soluble, and gaseous forms to give the total trace element content of soils and sediments. The *vertical gray area* shows the transition between available (bio-available, mobile and soluble) and non-available forms

immobile (biologically unavailable) forms, to mobile (bio-available) forms is the principal mission of biogeochemistry.

In general, an element is considered "trace" if it occurs in the environment at a total concentration <100 mg kg^{-1}, or <100 ppm. This is a geological classification and provides no prediction about its biohazardous nature. The trace element of greatest concern in the SJV has been Se, which is naturally occurring in the geologic parent material from the Coast Range, and routinely present in the soils and groundwaters on the Westside (Engberg et al. 1998). It becomes hazardous at concentrations much less than 100 ppm; for example, Se is a toxin in the aquatic environment at concentrations above 2 μg L^{-1} *i.e.* 2 ppb (Lemly 1993). In addition, the trace elements boron (B), arsenic (As), molybdenum (Mo), uranium (U), and vanadium (V) are frequently measured at elevated concentrations and, in some cases, could pose an environmental hazard or limit the reuse and discharge of agricultural wastewater. All of these elements occur naturally in the environment and enter the biogeochemical cycles through the weathering of soil minerals and soil-forming reactions. Human activities in the SJV have contributed to their enrichment in the groundwater, sediments, and agricultural drainage waters. In this chapter, we cover the chemical reactions that affect the solubility and bioavailability of these toxic trace elements in soils, sediments, and drainage waters. We focus on the inorganic chemical processes in soils and sediments. A later section in this chapter will discuss biochemical transformations and formation of organic compounds.

5.2 Sources and Sinks of Trace Elements – The Solid Phase

5.2.1 Dissolution and Precipitation of Solid Phases

Trace elements originate in soils and rocks as either primary mineral phases or occupying lattice spaces within more common minerals. The replacement of a trace element for a more abundant element in a mineral lattice (isomorphic substitution) is a common occurrence. Examples of isomorphic substitution include Se and As replacing S in pyrite (FeS_2), and B for Al in aluminosilicate minerals. Trace elements also occur in primary mineral phases such as arsenopyrite (FeAsS), molybdenite (MoS_2), and tourmaline ($Na(Fe,Mg)_3B_3Al_6Si_6O_{27}(OH)_4$). Exposure of these minerals to environmental conditions on the surface of the earth begins the weathering process. Many of these minerals are thermodynamically unstable in the presence of water at 20 °C, and so they dissolve. New, more stable minerals form, and the trace elements may remain in solution or precipitate as part of the new mineral assemblage. Minerals formed in soils as part of the weathering process are called secondary minerals and are dominated by the phyllosilicate clays (layered clays), hydrous metal oxides, carbonate minerals, and evaporite minerals. It is important to note that the weathering process is slow and affected by the amount of water in contact with the minerals and by soil temperature. In many cases, trace elements may be continuously supplied to the soil solution as a result of mineral weathering reactions. This has been shown for B in agricultural soils from the SJV; it is termed "boron regeneration" (Peryea et al. 1985). Thus, the problem of trace elements in soils and sediments may, in some cases, be an ongoing problem that will not be solved by simple "leaching" to remove the soluble contaminants. Although the original solid phase source of trace elements in soils and sediments is minerals, these elements can be associated with the organic fraction.

Soils on the west side of the SJV were irrigated initially with groundwater but additional development of the land occurred when the California Aqueduct was completed in the 1960s. Soils on the Westside were formed on alluvial materials from the Coast Range Mountains, which contain minerals rich in toxic trace elements. Rainfall, which amounts to less than 150 mm year^{-1} over most of the SJV, was supplemented by an additional 1,500 mm year^{-1} through irrigation. Irrigation water artificially increased the chemical weathering of soil minerals by increasing the amount of water in the soil. Water is applied during the hottest months of the year, which further accelerates dissolution reactions.

An example of a chemical weathering reaction that releases a trace element (boron, in this case) is the dissolution of tourmaline and the formation of a secondary phyllosilicate clay mineral, kaolinite (Eq. 5.1). Notice that the weathering reaction consumes acid, which in this case is carbonic acid formed from the CO_2 and water. Carbon dioxide is a product of respiration by crop roots and soil microbes, so irrigation provides both water and carbonic acid for the weathering reaction.

$$\begin{aligned}&Na(Fe, Mg)_3B_3Al_6Si_6O_{27}(OH)_4\ _{(tourmaline)} + 7CO_2 + 12H_2O \rightarrow \\ &Na^+ + 3(Fe^{2+},\ Mg^{2+}) + 7HCO_3^- + 3B(OH)_3^0 + 3Al_2Si_2O_5(OH)_4\ _{(kaolinite)}\end{aligned} \tag{5.1}$$

Again, Fig. 5.1 summarizes what we have discussed so far and provides a foundation for the discussion that follows. The total trace element concentration in soils and sediments is comprised of inorganic and organic forms found in solid, soluble, and gaseous phases. An adsorbed form is inserted between the solid and soluble forms because this is potentially a very bioavailable form. The adsorbed form is closely associated with solid phase surfaces but may be readily desorbed to a soluble form. It is also possible for gaseous forms to be associated directly with the solid phase, but this reaction is largely unrecognized for trace element gases. The broad, vertical band separating unavailable forms from the available forms of trace elements represents a range of biologically-active forms that depend on environmental conditions.

Trace elements may also be removed from solution by precipitation. The formation of secondary minerals in soils is a common process, and trace elements may be incorporated into the newly formed carbonate, phosphate, and silicate solid phases, or may precipitate as discrete phases, such as arsenopyrite (FeAsS) or elemental Se. For the trace elements of greatest concern in the SJV, precipitation occurs most often under reducing conditions, in which the elements are converted to lower oxidation states (Amrhein et al. 1993).

Trace elements can co-precipitate with evaporite minerals formed in drainage water evaporation ponds. Levy et al. (1994) found that V and U did not accumulate in evaporation pond minerals, if the alkalinity was >10 mmol $CaCO_3$ L^{-1}, but under low alkalinity, U partitioned into the evaporites. Ong et al. (1997) reported that Se, As, Mo, and B were largely excluded from the evaporite minerals formed in sulfate-rich wastewater ponds and demonstrated that the minerals formed would not be considered "hazardous waste." However, the brine solutions remaining after the evaporite minerals had precipitated were found to have high trace element concentrations that would classify these solutions as "hazardous waste" (Ong et al. 1997).

Both adsorption and precipitation can occur in soils, and it is often difficult to distinguish the controlling process or the resulting solid phases because of the low concentrations. It is sometimes difficult to distinguish between these two processes when trace elements are lost from solution; however, the difference can be extremely important for their management. If conditions exist for trace element removal by precipitation, then as long as those conditions remain, the trace element will be precipitated and accumulation will continue indefinitely. In the case of adsorption, accumulation of the trace element will continue only until the adsorption sites are filled; after that the trace element will remain in solution.

5.2.2 *Adsorption and Desorption Reactions*

Adsorption of trace elements onto soil minerals and organic matter is an important process that controls their solution concentrations. The strength of adsorption to a solid phase depends on the chemical properties of the trace element and the soil solution, and the surface properties of soil minerals and organic matter. Thus, solubility and bioavailability of these elements vary under different environmental conditions as indicated in Fig. 5.1. The hydrous metal oxides of iron (Fe), aluminum (Al), and manganese (Mn), and the broken edges of phyllosilicate clays are the most active adsorbers of oxyanions. The surface hydroxyl groups on soil minerals develop charge as a function of pH and ionic strength. The surface charge can be positive, negative, or uncharged. As pH decreases, positive charge increases and adsorption tends to increase. Positively-charged organic functional groups, like amine groups, also adsorb oxyanion trace elements.

The development of positive and negative surface charge on iron(III)-oxyhydroxides, like goethite (FeOOH), can be represented as follows:

$$\text{FeO-OH(s)} + \text{H}^+ \leftrightarrow \text{FeO-OH}_2{}^+\text{(s)} \tag{5.2}$$

$$\text{FeO-OH(s)} + \text{OH}^- \leftrightarrow \text{FeO-O}^-\text{(s)} + \text{H}_2\text{O} \tag{5.3}$$

where the FeO-OH indicates a hydroxyl group on the goethite solid surface exposed to the soil solution. The amount of surface charge is controlled by the pH of the soil solution; anions are attracted to the positively-charged sites, while cations are adsorbed to negatively-charged sites. Trace elements can be adsorbed onto the positively charged sites by two mechanisms: The first is a simple electrostatic attraction between the trace element oxyanion and the protonated hydroxyl group. This is called "*outer-sphere adsorption*" and can be represented with selenite adsorption as follows:

$$\text{FeO-OH}_2{}^+\text{(s)} + \text{SeO}_3{}^{2-} \leftrightarrow \text{FeO-OH}_2\text{OSeO}_2{}^{1-} \tag{5.4}$$

If two adsorption sites are close together on the surface, the anion may be held by electrostatic attraction to both sites and a bidentate bond is formed. In this example with selenite ($SeO_3{}^{2-}$), the strength of bonding to goethite is relatively weak, resulting in a limited decrease in selenite bioavailability and mobility.

A second adsorption mechanism involves replacement of the surface hydroxyl (one or more) by the O^{2-} of the oxyanion, and the oxyanion becomes part of the surface structure of the mineral. This type of bonding is called *chemi-sorption, ligand exchange, inner-sphere adsorption, or specific adsorption*. This type of chemical bond is substantially stronger than a simple electrostatic adsorption. An example of specific adsorption using selenite is:

$$\text{FeO-OH}_2{}^+\text{(s)} + \text{SeO}_3{}^{2-} \leftrightarrow \text{FeO-OSeO}_2{}^{1-} + \text{H}_2\text{O} \tag{5.5}$$

Specific adsorption occurs most strongly with selenite ($SeO_3{}^{2-}$), arsenate ($AsO_4{}^{3-}$), arsenite ($As(OH)_3{}^0$), borate ($B(OH)_4{}^-$, molybdate ($MoO_4{}^{2-}$), and vanadate ($VO_4{}^{3-}$). With this adsorption mechanism, the anions would have a greatly decreased bioavailability and mobility. In contrast, outer-sphere adsorption is more common with selenate ($SeO_4{}^{2-}$), vanadyl ($V(OH)_3{}^0$), and the uranium complexes ($UO_2{}^{2+}$, $UO_2(CO_3)_2{}^{2-}$, $UO_2(OH)_3{}^-$).

Adsorption reactions are affected by ionic strength, pH, competing ions, and the presence of complexing species. Most trace element oxyanions act as weak acids, and can readily gain or release H^+ ions from their coordinated O^{2-} ions. For example, as pH decreases $AsO_4{}^{3-}$ is converted sequentially into $HAsO_4{}^{2-}$, $H_2AsO_4{}^{1-}$, and $H_3AsO_4{}^0$. The relationship among the chemical species can be written as follows, with the equilibrium constants, K_1, K_2, and K_3 called the first, second, and third dissociation constants for arsenic acid:

$$H_3AsO_4{}^0 \leftrightarrow H_2AsO_4{}^{1-} + H^+ \quad K_1 = 10^{-2.3} \tag{5.6}$$

$$H_2AsO_4{}^{1-} \leftrightarrow HAsO_4{}^{2-} + H^+ \quad K_2 = 10^{-6.8} \tag{5.7}$$

$$HAsO_4{}^{2-} \leftrightarrow AsO_4{}^{3-} + H^+ \quad K_3 = 10^{-11.6} \tag{5.8}$$

Thus, the pH strongly affects the reactivity and mobility of most oxyanions.

Other oxyanions, hydroxyl ions, and fluoride ions will compete for adsorption sites occupied by trace element oxyanions. This phenomenon is called *competitive anion exchange* and it affects the mobility and bioavailability of toxic trace elements in soils. For example, phosphate ($HPO_4{}^{3-}$) will compete for adsorption sites occupied by selenite on goethite as follows:

$$\text{FeO-OSeO}_2{}^- + HPO_4{}^{2-} \leftrightarrow \text{FeO-OPO}_2(OH)^- + SeO_3{}^{2-} \tag{5.9}$$

The competitive exchange may displace both inner-sphere and outer-sphere ions. There has been extensive work on the modeling of adsorption and competitive anion exchange reactions in soils (Glasaur et al. 1995; Duff and Amrhein 1996; Fox and Doner 2002a; Gao et al. 2004, 2006). The most popular adsorption model used to describe the interaction of adsorption sites, oxyanions, pH, and ionic strength has been the "constant capacitance model" (Manning and Goldberg 1996a, b; Goldberg et al. 2000, 2002, 2005a, b; Fox and Doner 2002a). Fox and Doner (2002b) found that goethite-coated sand was a strong adsorber of As, Mo, and V in agricultural drain water, while calcite adsorbed no As and only low amounts of Mo and V, indicating that specific adsorption onto metal-oxides is an important process in soils.

5.2.3 *Soluble Forms of Trace Elements (Speciation)*

The available forms of the trace elements of concern in the SJV typically occur as oxyanions, or as uncharged species, which affect their biogeochemistry. The important oxyanionic forms of these elements are selenate ($SeO_4{}^{2-}$), selenite ($SeO_3{}^{2-}$), arsenate ($AsO_4{}^{3-}$), arsenite ($As(OH)_3{}^0$), borate ($B(OH)_4{}^-$), boric acid ($B(OH)_3{}^0$), molybdate ($MoO_4{}^{2-}$), uranyl carbonate ($UO_2(CO_3)_2{}^{2-}$), and vanadate ($VO_4{}^{3-}$). The tendency for an element to form oxyanions is based on an element's ionization potential, which is the ratio of the ionic charge on the ion to its ionic radius (Table 5.1). When an element has a high positive charge (cation) and a

Table 5.1 Ionization potentials (ion charge/ionic radius) for elements found in ground water, soil solutions, and rivers

Element	Ion charge under oxidizing condition	Ion radius (nm)	Ionization potential (nm^{-1})	Category
Se	+6	0.042	143	Trace element oxyanions
B	+3	0.023	130	Trace element oxyanions
As	+5	0.046	109	Trace element oxyanions
Mo	+6	0.062	97	Trace element oxyanions
V	+5	0.059	85	Trace element oxyanions
U	+6	0.080	75	Trace element oxyanions
P	+5	0.035	143	Trace element oxyanions
N	+5	0.013	385	Major element oxyanions
S	+6	0.030	200	Major element oxyanions
C	+4	0.016	250	Major element oxyanions
Si	+4	0.042	95	Major element oxyanions
Cl	−1	0.181	−6	Non-oxyanion elements
Na	+1	0.097	10	Non-oxyanion elements
Ca	+2	0.099	20	Non-oxyanion elements
Mg	+2	0.066	30	Non-oxyanion elements
K	+1	0.133	8	Non-oxyanion elements
Al	+3	0.051	59	Non-oxyanion elements

relatively small ionic radius, it will electrostatically coordinate H_2O and OH^- and repel H^+ ions, thereby leaving O^{2-} ions bonded to the cation. If an element has a low positive charge, a negative charge, or a large ionic radius, it is unlikely to form an oxyanion. Aluminum (III) and Si (IV) can be considered the borderline elements because they coordinate OH^- ions but do not have high enough ionization potentials to repel H^+ from the coordinated OH^-. Uranium forms strong complexes with both carbonate and hydroxyl ions, converting the uranyl ion (UO_2^{2+}) into an anion in water and soils (Duff and Amrhein 1996).

In many cases, the oxidation states of trace elements are lower in the primary minerals because these minerals are often formed under reducing conditions, high temperatures, and high pressures. The weathering reactions involve dissolution of the lattice structure and may involve oxidation, if O_2 is present. An example of a chemical weathering reaction that requires O_2 is the oxidation of arsenopyrite (FeAsS). In this reaction (Eq. 5.10), the secondary mineral goethite is formed and all three elements in the arsenopyrite are oxidized.

$$\begin{aligned} FeAsS_{(arsenopyrite)} + 7/2O_2 + 3H_2O \rightarrow FeOOH_{(goethite)} + AsO_4^{3-} \\ + SO_4^{2-} + 5H^+ \end{aligned} \tag{5.10}$$

Agricultural soils are typically aerobic environments and this favors the oxidation of trace elements. However, the oxidation of reduced, mineral-bound trace elements may be quite slow and as a result, there is a gradual release of soluble toxic ions to the soil solution (Zawislanski and Zavarin 1996).

The solubility, speciation, bioavailability, and toxicity of Se, As, U, V, and Mo are controlled by the redox status of the soil, sediment, or groundwater. Reduced forms of these elements react with oxygen and become oxidized species. The most common reduced species of these elements are Se (IV, 0, −II), As (III, 0), V (III and IV), U (IV), and Mo (IV). The reduced forms of Se, U, V, and Mo are generally less soluble and less toxic than the oxidized forms because the lower oxidation state species form solid precipitates with low solubility. Arsenic (III), on the other hand, tends to be more toxic and mobile than the oxidized form, As (V).

Reducing conditions can occur in soils due to flooding or waterlogging, where O_2 diffusion into the soil is restricted. The presence of readily-decomposable organic matter will intensify the reducing conditions (lower E_H) and convert oxidized trace elements to their reduced forms. For example, under anaerobic conditions, Se will be reduced to zero-valent Se (Se^0), which precipitates as a red-colored solid (Losi and Frankenberger 1998). If the E_H is sufficiently low, sulfate reduction will occur, and sulfide minerals can form. Thus, Se and As can be co-precipitated into iron (II) sulfide phases under anaerobic conditions. Under moderately reducing conditions, selenate (SeO_4^{2-}) and arsenate (AsO_4^{3-}) will be reduced to selenite (SeO_3^{2-}) and arsenite ($As(OH)_3^0$), respectively. Selenite interacts with soil minerals via adsorption reactions more strongly than selenate (SeO_4^{2-}), but arsenite, mentioned above, is more soluble and less strongly adsorbed than arsenate (AsO_4^{3-}).

5.2.4 Volatilization Reactions

Selenium and As can be converted to gaseous forms via microbial processes of methylation (Chasteen 1998) or volatilized through plants. Dimethyl selenide (CH_3SeCH_3), dimethyl diselenide ($CH_3SeSeCH_3$), dimethyl selenenyl sulfide (CH_3SeSCH_3), methaneselenol (CH_3SeH), and hydrogen selenide (H_2Se) are the volatile Se species formed in soils and plants (Chasteen 1998). Microbial reactions produce arsine (AsH_3), methylarsine (CH_3AsH_2), dimethylarsine ($(CH_3)_2AsH$), and trimethylarsine ($(CH_3)_3As$), which transfer As in the soil to the atmosphere. Extensive work has been undertaken to evaluate the factors that affect Se and As volatilization with the hopes of developing bioremediation strategies for these contaminants. Arsenic and Se volatilization may be important pathways by which these elements are removed from soils, depending on their concentration in the soil, soil moisture content, kinds of soil amendments applied, types of vegetation, and other environmental factors (Bañuelos et al. 2005; Thompson et al. 2003; Frankenberger and Arshad 2002; Karlson and Frankenberger 1989).

5.3 Reactions of Specific Trace Elements

We have discussed some general soil reactions of various elements of interest as examples of soil chemical processes. Here, we focus on specific trace elements and important results of research undertaken as part of the UC Salinity Drainage Program.

5.3.1 Selenium

5.3.1.1 Distribution

One of the primary questions posed when Se was discovered as an environmental contaminant of the agricultural drainage water in SJV was, "What are the chemical forms of Se in the soils in the irrigated area?" There were two distinct approaches to answering this question. One approach was to chemically fractionate soil and find the Se distribution in various soluble fractions. The second approach was to examine the reactions of the various soluble chemical species of Se to see what important soil components are responsible for its adsorption or precipitation.

Agricultural soils of the Westside do not typically contain enough total Se to be considered seleniferous. Seleniferous soils usually have total Se concentrations in excess of 2–4 mg kg^{-1}. In applying the first approach to chemically speciate Se, this low concentration requires very careful analytical procedures. Abrams and Burau (1989) studied a Ciervo clay soil from the Panoche fan area of the SJV using a chemical fractionation method. They found that 20 % of the total Se was in the organic matter fraction of the soil. Also, they found that in the organic fraction that selenomethionine, an amino acid in which Se(−II) substitutes for sulfur, was one of the components. The remaining 80 % of the Se was associated with inorganic components, which were not identified. This finding has importance relative to the time scale for releasing Se from an unavailable form to an available form. We would expect the introduction of irrigation water into the soils, especially in hot weather, to result in an increase in soil organic matter mineralization and the release of Se to available forms. Lipton (1991) developed a more detailed chemical fractionation method for Se. He found the total Se concentration in two important agricultural soils from the Panoche fan of the SJV to be 990 and 840 μg Se kg^{-1} for the Ciervo and Panoche soils, respectively (Table 5.2). This result indicated that these soils were not seleniferous, by definition, but could still be an important Se source in drainage waters. In fact, drainage waters from this area were considered an important source of Se-rich water sent to the Kesterson Reservoir in the 1980s. As discussed above, the total Se concentration may not be a good indicator of bioavailable Se. In an effort to determine the forms of Se in soils and sediments, various chemical extraction procedures were developed to partition Se into soluble, adsorbed, and solid phase fractions (Lipton 1991; Martens and Suarez 1997; Wright et al. 2003). These methods rely on selecting a series of chemical extractants that start with a very mild extraction and end with a very harsh chemical extraction. Each extractant is selected to dissolve or exchange different phases of Se. Although how specific each extractant is for a particular form of Se may be subject to question, the methods Wright et al. (2003) developed do provide valuable insights into the geochemistry of an element. In addition, predictions can be made about the potential availability of an element (Se, in this case) based on its relative ease or difficulty of extraction.

Table 5.3 shows the distribution of Se in the Ciervo and Panoche soils and in sediment samples from Kesterson Reservoir. The total Se concentration of the

Table 5.2 Some chemical and physical properties of two agricultural soils (Ciervo and Panoche, 0–15 cm depth) from the Westside of San Joaquin Valley (Lipton 1991)

Soil	Classification	Total Se (μg kg^{-1})	pH (1:1)	EC (dS m^{-1})	CEC ($cmol_c$ kg^{-1})	Total organic C (g kg^{-1})
Ciervo clay loam	Typic Torriorthent	990	8.6	2.0	34	11.8
Panoche sandy loam	Typic Torriorthent	840	8.2	22.2	19	4.8

Table 5.3 The distribution of selenium in soils and Kesterson sediment among various extractable forms

	Soluble		Adsorbed		SOM	Minerals	
	Se_{total}	Se(IV)[a]	Se_{total}	Se(IV)[a]	Se_{total}	Active Se_{total}	Inactive Se_{total}
	μg kg^{-1}						
Ciervo clay loam[b]	36	9	29	21	257	206	418
Panoche sandy loam[b]	21	9	31	22	171	107	552
Kesterson sediment[c]	298	68	417	269	4,341	240	<72
Species distribution	Soluble forms		Adsorbed forms		Solid forms		

Selenium from minerals potentially more sensitive to environmental changes are indicated by Se_{total}, Active while more resistant mineral sources are indicated by Se_{total}, Inactive. The schematic diagram above the table corresponds to Fig. 5.1

[a]Amount of Se(IV) in the total Se

[b]Lipton (1991)

[c]Tokunaga et al. (1991)

Ciervo soil was 990 μg kg^{-1}, divided between the soluble, adsorbed, and solid phases. In the soluble fraction of the Ciervo soil, Lipton (1991) found a total soluble Se concentration of 36 μg kg^{-1} soil, or about 4 % of the total Se in the soil. Of the 36 μg $Se_{soluble}$ kg^{-1}, 9 μg Se kg^{-1} was in the form of Se (IV). The remaining 27 μg kg^{-1} of the soluble Se was most probably the selenate species (Se VI). About 3 % of all forms of soil Se were readily available to leaching water, in the case of the Panoche soil, and about 6 % for the Kesterson sediment (Tokunaga et al. 1991). However, in the case of the Kesterson sediment where the total Se concentration was nearly 5,400 μg Se kg^{-1}, the soluble component was approximately 300 μg Se kg^{-1}. Leaching of the Kesterson sediment would have released very high concentrations of soluble Se into the soil solution. When we consider the Se from soils and sediments in soluble forms, it is important to understand the time scale for the release. Commonly, this is the amount of Se released to solution over a period of a few hours. In a field situation, slow releases from other forms may continue to add Se to the mobile phase over time.

Table 5.4 Distribution of Se in Panoche soil, Ciervo soil, and Kesterson sediment using the Lipton chemical fractionation method (Lipton 1991)

				Mineral (%)	
Source	Soluble (%)	Adsorbed (%)	Soil organic matter (%)	Active	Inactive
Kesterson sediment	5.55	7.77	80.87	4.47	1.34
Panoche sandy loam	2.38	3.51	19.39	12.13	62.59
Ciervo clay loam	3.81	3.07	27.17	21.78	41.19

The adsorbed form of Se is found by extracting the soil or sediment with a solution containing a competitive anion, such as phosphate ($HPO_4{}^{2-}$), to exchange with adsorbed selenite ($SeO_3{}^{2-}$) e.g. Eq. 5.9 and selenate ($SeO_4{}^{2-}$). We can see from the results in Table 5.4 that adsorbed forms account for 3–8 % of the total Se in the two soils and sediment. Selenite accounted for the largest fraction of Se desorbed. This finding was in agreement with numerous studies showing selenite is adsorbed with greater strength than selenate to soils (Goldberg and Glaubig 1988; Neal and Sposito 1989; Neal et al. 1987). What does this mean in terms of managing these soils and sediments? The adsorbed form is a slow-release source for soluble Se. Also, changes in management practices that involve addition of competitive anions, such as phosphate, may result in a more rapid release of Se from the solid phase to solution.

The selenium retained in soil organic matter accounted for 19–27 % of the total soil Se (Table 5.4; Lipton 1991). In contrast, the organic matter component of the Kesterson sediment retained more than 80 % of the total Se or nearly three times as much on a percentage basis (Tokunaga et al. 1991). This means that any changes in soil or sediment conditions that might increase the rate of organic matter mineralization, such as aeration or increased microbiological activity, would potentially make this pool of Se mobile and bioavailable. Another important question was why did these soils have much less Se associated with the organic fraction than the Kesterson sediment? The answer seems to be that the origin of the Se in the soils was from *in situ* mineral weathering, while the major Se source for the Kesterson sediment was from agricultural drainage water that was being evapo-concentrated in the ponds at Kesterson. For the soils, there were relatively small amounts of Se available for incorporation into the terrestrial organic matter produced on site. In the Kesterson Reservoir, the concentrations of soluble Se were higher and the bioaccumulation of Se by algae resulted in a sediment surface layer rich in organic Se.

Lipton (1991) separated the Se extracted by the harsher chemical treatments into "active" and "inactive" mineral components. The active mineral components represent minerals that may be more readily dissolved by changes in acidity or redox conditions in the soil environment. For example, soil carbonates are readily dissolved under acid conditions and manganese oxides dissolve quickly under reducing conditions. From 12 to 22 % of the total Se in the Ciervo and Panoche soils was associated with the active mineral fraction, but only 5 % in the Kesterson sediment. As with the Se associated with organic matter mineralization, these results indicate there will be a slow release of Se to potentially available forms over a long time scale, i.e., perhaps many years. Assessment of Se in "inactive" minerals requires very harsh chemical treatments, such as boiling the soil material in strong acid and base solutions. From about 40 to 60 % of the total Se, or from 418 to 552 μg Se kg^{-1}, was extracted from the Ciervo and Panoche soils with this treatment (Table 5.3). Only about 1 % of the total Se, or <72 μg Se kg^{-1}, was extracted from the Kesterson sediment using the same treatment (Table 5.3). Selenium from the "inactive" minerals is thought to be very slowly released over long time scales measured in hundreds of years. Normally, this Se would not be an important component of the bioavailable Se but irrigation does increase the rate of mineral weathering and the release of refractory forms.

5.3.1.2 Reaction Mechanisms and Models

Earlier research identified some of the important soil minerals that adsorb Se anions (Geering et al. 1968; Cary and Allaway 1969; Hingston et al. 1971, 1972, 1974), and helped to identify which chemical species were selectively adsorbed. Besides the iron and aluminum oxyhydroxide minerals forming strong surface bonds with selenite, there was some evidence presented by Singh et al. (1981) and Tokunaga et al. (1991) that soil carbonate may also contribute to Se retention. Goldberg and Glaubig (1988) in their chemical modeling study found that calcareous minerals in soils play an important role in selenite sorption. The role of calcium carbonate (calcite) in Se sorption was later reviewed by Doner and Zavarin (1997). They did not show clear evidence of selenate or selenite incorporation into calcite; however, Reeder et al. (1994) found that selenate could substitute for carbonate in that mineral. In general, the current evidence is that calcium carbonate can retain Se as selenite and selenate, but there is some uncertainty as to the exact mechanism that occurs in nature (Zavarin 1999).

Neal et al. (1987) found that selenite was adsorbed to alluvial soils from the Westside of the SJV. On the other hand, in a later investigation of the same soils, they found that selenate was not adsorbed (Neal and Sposito 1989). Goldberg and Glaubig (1988) found that both kaolinite and montmorillonite, two important clay minerals, could adsorb selenite but not selenate. These findings clearly show that selenate is the more mobile species in soils. Recently, (Pizzini 2007) found that selenate adsorption could be found in soils when Se concentrations in solution were in the 5–50 μg L^{-1} range. The amount of adsorption was directly related to the iron oxide content of the soil.

The approaches for studying the chemistry of soil Se discussed above do not provide direct evidence for particular reactions (specific adsorption versus precipitation) or the associated mineral phases where the Se occurs in native soils. Using x-ray absorption microprobe spectrometry provides a method for identifying elemental composition, oxidation state, and element distribution on a micrometer scale. Resolving distributions on the micrometer scale is essential, given the fine grain, heterogeneous nature of soils. Using this technique on undisturbed, native soils from the Panoche Hills (source material from Panoche fan soils), Strawn et al. (2002) were able to identify selenite and selenate species and their associated minerals. This was the first time direct evidence in soil was found for the association of Se with iron (Fe) minerals, such as ferrihydrite, goethite, and jarosite. These results showed that both selenite and selenate were associated with Fe minerals, with selenite being the dominant Se form.

5.3.2 Arsenic

Research on the possible use of constructed wetlands to decrease Se concentration in agricultural drainage water in the southern SJV was conducted over the period of a few years. Besides Se, other trace elements were considered as part of that project, namely As, Mo, and V. Our focus here is on As. Molybdenum and V will be discussed in the following sections. As discussed, soluble As is normally in either a V or III oxidation state. Arsenite (As^{3+}) is usually considered the more toxic and mobile form, while arsenate (As^{5+}) readily bonds to iron and aluminum minerals. Soils in this study were submerged and the pH of the soil environment was alkaline, ranging between 7.5 and 10. Figure 5.2 shows the distribution of As according to extractable species with depth of soil in a constructed wetland.

The total extractable As pre-flood was 13.2 mg As kg^{-1} and after 2 years of flooding, the total was < 10 mg kg^{-1} at the surface and decreased to <2 mg kg^{-1} with depth (Fig. 5.2). On the other hand, water-soluble and phosphate-extractable forms of As increased in concentration in most cases. We can conclude that the soil, under conditions of continuous flooding, was not an accumulator of As, even though As was continuously added in the flood water. In fact, flooding of the soil with these drainage waters seems to be removing As from the system, especially at the deeper soil depths. The reason for this spatial redistribution of As is not well understood, although conversion to organic and gaseous forms may partly explain the loss. The redox environment of a flooded soil is complex, although under anaerobic conditions, arsenite, commonly considered the more mobile and more toxic form of As ($As(OH)_3{}^0$), is expected to form. However, this may not be true in more alkaline environments, which was the condition in these ponds. Sun and Doner (1998) found that the adsorption of arsenite on goethite, a common soil mineral, is much greater than arsenate in the alkaline pH range. In the neutral and acid pH range, arsenate is the predominant species adsorbed. Again, the first direct evidence for this finding in a native soil was collected by Strawn et al. (2002) in their x-ray absorption investigation using x-ray spectrophotometry techniques. In their studies, they were able to identify the molecular bonding mechanism of arsenite and arsenate using infrared spectroscopy (Sun and Doner 1996, 1998; Sun et al. 1999). Understanding bonding mechanisms helps in our predictions about which molecular forms of As react with minerals and the conditions for those reactions. Although biological redox reactions are very important in soil systems, their studies also illustrated the importance of manganese minerals in inducing redox reactions with As.

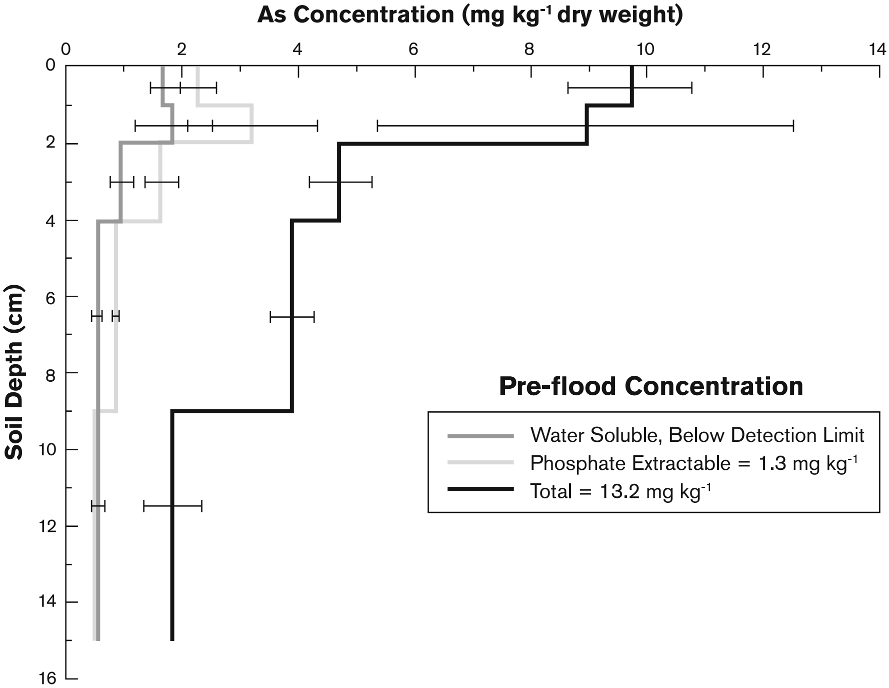

Fig. 5.2 Concentration of As using different chemical extractants at different soil depths in a constructed wetland after 2 years under flooded conditions. Pre-flood conc. Show results of bulk soil sample analysis before flooding (Fox and Doner 2003)

5.3.3 Molybdenum

Molybdenum (Mo) is an essential element for both plants and animals but high levels in herbage can cause molybdenosis, a Mo-induced Cu deficiency in livestock. There is some indication that Mo, along with Se, can cause avian embryotoxicity at concentrations found in bird eggs collected from evaporation ponds in the SJV (Ohlendorf et al. 1993). The most common form is molybdate (MoO_4^{2-}), and with a pK_a of 4.2, the bimolybdate ion ($HMoO_4^{-}$) would only be significant in acidic waters. A study of mineral formation during the evaporative concentration of agricultural wastewater suggested that Mo might precipitate as $CaMoO_4$ (Levy et al. 1994). While molybdate is not strongly adsorbed on metal oxides via specific adsorption, it does have an affinity for organic matter and has been shown to coordinate with the pyrrole groups in porphyrins. Ong and Tanji (1993) postulated that organic-Mo interactions might explain the behavior of Mo in evaporation ponds.

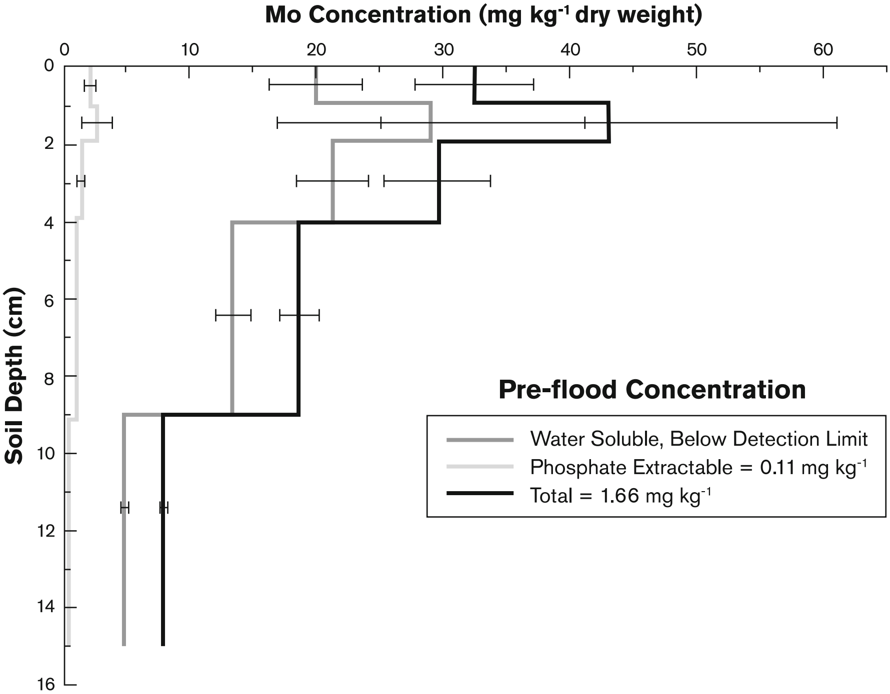

Fig. 5.3 Concentration of Mo using different chemical extractants at different soil depths in a constructed wetland after 2 years under flooded conditions. Pre-flood conc. show results of bulk soil samples before flooding (Fox and Doner 2003)

Under low redox conditions, molybdate can be reduced to Mo^{4+}, which readily precipitates as MoS_2 (molybdenite) or co-precipitates with iron(II)-sulfide minerals, such as pyrite ($Fe_{1\text{-}x}Mo_xS_2$). This reaction can be illustrated as follows:

$$MoO_4{}^{2-} + 6H^+ + 2SO_4{}^{2-} \rightarrow MoS_2 + 4.5O_2 + 3H_2O \qquad (5.11)$$

In this reaction, both Mo^{6+} and S^{6+} are reduced to Mo^{4+} and S^{2-} to form the precipitate, MoS_2. In a study on the use of constructed wetlands for remediation of agricultural drainage water, Fox and Doner (2002a, b, 2003) reported extraordinarily large accumulations of Mo within 4–10 cm of the sediment surface. Some of the results of this work are shown in Fig. 5.3.

The pre-flood concentration of Mo was 1.66 mg kg^{-1} but after applying drainage water to the soil for 2 years the total Mo concentration was more than 40 mg kg^{-1}. This is more than a 20-fold increase in concentration, and nearly all of it as soluble Mo. The high concentrations of soluble Mo may have been an artifact of the sampling method. As long as the soil remained anoxic in the wetland, the concentration of soluble Mo most likely remained low. However, as soon as the soil was exposed to oxic conditions with collection of the sample, oxidation caused an increase in the soluble Mo concentration. There are two possible reactions that can account for the changes in Mo solubility. One pathway is the oxidation of MoS_2 to MoO_4^{2-} and SO_4^{2-} when the sediment is exposed to air (Amrhein et al. 1993). The second pathway is formation of thiomolybdate during reduction (Eq. 5.12). In this reaction, Mo^{6+} is not reduced to the +4 oxidation state; instead, it remains as VI and only sulfate is reduced to S^{2-} (sulfide). Preliminary studies (Fox 2000; Fox and Doner 2002b) using XANES indicated that thiomolybdate was formed under anoxic conditions, in addition to the solid-phase molybdenite (MoS_2).

$$MoO_4^{2-} + 8H^+ + 4SO_4^{2-} \rightarrow MoS_4^{2-} + 4H_2O + 8O_2 \tag{5.12}$$

In this equation, sulfide replaces all the O^{2-} coordinated with Mo to form tetrathiomolybdate, a soluble anion. Other forms of thiomolybdate are known, e.g., $MoOS_3^{2-}$ or trithiomolybdate. When exposed to oxygen, anaerobic soils or pond sediments released Mo to solution, since all of the reduced forms are readily oxidizable (Amrhein et al. 1993; Fox 2000; Fox and Doner 2002b, 2003).

5.3.4 Uranium

High concentrations of U have been measured in agricultural evaporation pond water (up to 22 mg U L^{-1}) and in pond sediments (290 mg U kg^{-1}) (Bradford et al. 1990; Chilcott et al. 1990; Duff et al. 1997a). Although most of the avian toxicities in the SJV have been associated with elevated levels of Se, studies have shown that U may be involved in waterfowl problems in the valley (Ohlendorf et al. 1993). In the late 1980s, more than 7,300 ha (18,039 acres) of evaporation ponds existed in the valley and most of these ponds had U concentrations that exceeded the USEPA drinking water standard of 0.02 mg U L^{-1} (Chilcott et al. 1990).

Uranium exists in nature in three oxidation states: U(IV), U(V), and U(VI). The most soluble form is U(VI) which exists as the ion UO_2^{2+} (uranyl), and ion pairs of this species. Uranium V is unstable due to disproportionation and is not expected to dominate redox speciation in soils and waters. Uranium IV is sparingly soluble, forming $UO_{2(s)}$ (uraninite) under intense anaerobic conditions where sulfide concentrations are high (Duff et al. 1999). The uranyl ion forms strong complexes with OH^- and CO_3^{2-} resulting in $(UO_2)_2CO_3(OH)_3^{1-}$, $UO_2CO_3^0$, $UO_2(CO_3)_2^{2-}$, $UO_2(CO_3)_3^{4-}$ under oxic conditions.

The adsorption studies of dissolved U(VI) on soil surfaces, clays, carbonate minerals, and metal oxides have shown that the complexation with CO_3^{2-} lowers adsorption (Duff and Amrhein 1996). In waters with bicarbonate concentrations greater than 2 mmol L^{-1} and saturated with calcite, uranyl adsorption in calcareous SJV soils was low under both low and high carbon dioxide pressures. Chemical

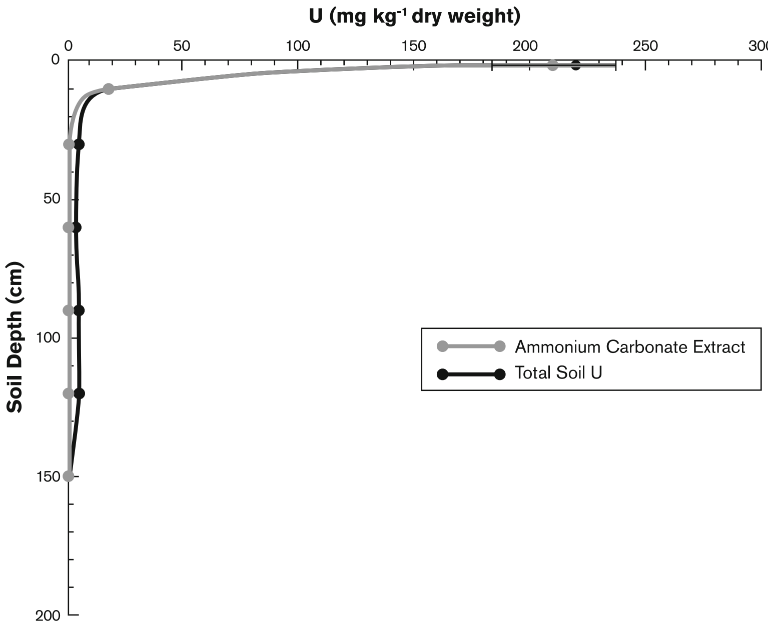

Fig. 5.4 Concentration of U in San Joaquin Valley evaporation pond sediments and subsurface soil using a 0.5 M $(NH_4)_2CO_3$ extractant and concentrated nitric acid digestion (Duff et al. 1997b)

speciation calculations suggested that the mono-, di-, and tri-carbonate complexes of uranyl dominated in solution and, due to their charge and size, could not compete effectively for specific adsorption sites.

It was previously believed that under low oxygen conditions, U(VI) was always converted to U(IV), which would precipitate as $UO_{2(s)}$ (uraninite). However, Duff et al. (1997a) reported that pond sediment U was approximately 25 % U(IV) and 75 % U(VI), a ratio that corresponds closely to that of pitchblende ($U_3O_{8(s)}$). This mix-oxidation-state solid was highly soluble when exposed to air and soluble in ammonium carbonate, suggesting that the abandoned, dry evaporation ponds could be a source of concentrated, soluble U (Fig. 5.4).

Further studies indicated that $UO_{2(s)}$ was formed only under intense SO_4^{2-} reducing conditions, and the reduction occurred by chemical reduction via the sulfide ion (Duff et al. 1999). Additional studies in the lab with the alga *Chlorella* and filamentous algae collected from evaporation ponds in the SJV showed that the mechanism of U accumulation in the sediments involved algal bioaccumulation (Duff et al. 1997b, 1999). Thus, the very high concentrations seen in the surface sediments (Fig. 5.4) are attributed to the deposition of dead algae.

The uranium in these sediments has the natural abundance of isotopes and it might be economical to collect the surface sediments for the ^{235}U. The lower limit for U ore is considered 300 mg kg^{-1} and the average surface sediments from several SJV evaporation ponds was 200 mg kg^{-1}. However, the incentive for site remediation and the ease of access to surface sediments might make recovery a viable option.

5.3.5 Vanadium

Vanadium is an essential micronutrient involved in lipid metabolism in higher animals and an essential element for some green algae and marine microalgae (Macara 1980; Jones et al. 1990). Vanadium is a potent inhibitor of the "sodium pump" or Na-K-ATPase activity, and may affect the salt gland in shore birds (Macara 1980).

Vanadium can exist in three valence states in soils and pond sediments: III, IV, and V. The most oxidized form, vanadate, has a high ionic potential, forms strong hydrolysis species, and behaves much like phosphate. Vanadate is the most soluble of the three oxidation states which leads to its accumulation in agricultural drainage water. The hydrolysis species of vanadate are: $H_3VO_4^0$, $H_2VO_4^{1-}$, HVO_4^{2-}, and VO_4^{3-}, with the −1 and −2 species dominant in soil solutions. The chemical similarity between vanadate and phosphate suggests that vanadate can inhibit reactions involving phosphate (Willsky 1990). Under anaerobic conditions, the vanadyl ion (VO^{2+}) is dominant. Vanadyl is strongly complexed with organic matter (much like Cu^{2+}), and vanadyl was identified, using electron spin resonance, in the organic layers formed on the bottom of agricultural wastewater evaporation ponds in the SJV. Laboratory and field studies have shown that vanadium partitions to the sediments of agricultural wastewater evaporation ponds under low redox conditions (Amrhein et al. 1993). If exposed to aerobic conditions, the sediment-bound V is resolubilized, apparently due to oxidation of vanadyl back to vanadate (Amrhein et al. 1993). Based on evaporative concentration studies of agricultural wastewater, Levy et al. (1994) concluded that V is not lost to the pond sediments via precipitation or coprecipitation with evaporite minerals.

Fox and Doner (2002b, 2003) studied the behavior of V in constructed wetlands flooded with agricultural drainage water. They found V accumulation and losses to be dependent on reducing conditions. In moderately reducing conditions in the field, V may be mobilized. In wetlands with a greater water column over the sediment surface and, ostensibly greater reducing conditions, some V accumulation appeared to occur in the sediments.

5.3.6 Boron

Boron is an essential element for plant growth, but becomes phytotoxic at concentrations only slightly above those considered essential. For example, irrigation water should have > 0.02 mg B L^{-1} to ensure adequate soil B levels; hydroponic solutions designed for optimum plant growth should have 0.25 mg B L^{-1}; but irrigation waters with > 0.5 mg B L^{-1} will cause a reduction in yield of "boron sensitive" crops (Maas 1990; WPHA 2002). In many instances, the reuse potential of agriculture wastewater from the SJV may be limited by its B concentration, rather than total salinity.

Boron is an uncharged ion at most soil solution pH values, ($B(OH)_3{}^0$ or $H_3BO_3{}^0$), and is weakly adsorbed via specific adsorption on metal-oxides, most likely as borate ($B(OH)_4{}^{1-}$) (Goldberg et al. 2000, 2005a). In addition, there has been earlier evidence that magnesium (Mg) minerals may be involved in B retention (Rhoades et al. 1970). A recent study by Pittiglio (2003) demonstrated that Mg minerals could adsorb B but significant adsorption occurred only at pH ranges between 8.5 and 11.2. Only in the most alkaline soils might this mechanism of B retention be important. Boron is not redox-sensitive and concentrates by evapotranspiration of soil solutions and by evaporation in open ponds. Ong and Tanji (1993) found that evapoconcentration factors based on chloride accurately predicted B concentration in evaporation pond waters, indicating the non-reactivity of B towards precipitation or adsorption.

Boron concentrations have been observed to increase in solution when soils or pond sediments are subject to anaerobic conditions (Amrhein et al. 1993). This increase in B was attributed to the reductive dissolution of iron (III)-oxides and manganese oxides, and the subsequent solubilization of adsorbed B.

5.4 Summary

- The major form of water-soluble Se in soils was as selenate.
- The water-soluble Se constitutes only a small fraction of the total soil Se.
- Organic Se forms a large fraction of the total soil Se. Its exact molecular form or forms is largely unidentified.
- Under conditions of Se accumulation as found at Kesterson, organic Se accounts for the major fraction of total Se. Changing management methods that result in organic matter decomposition will most likely result in increased Se availability.
- Arsenic did not accumulate under anoxic conditions, as found at the constructed wetlands. There was some evidence of mobilization.
- Molybdenum under anoxic conditions accumulates most likely in some form involving sulfide. Changing from an anoxic to an oxic environment results in rapid and relatively large increases in the amount of soluble Mo.
- Uranium accumulates in evaporation pond sediments as a mix-oxidation-state solid, which is highly labile when exposed to oxygen. The relatively high alkalinity of the SJV waters keeps the U in solution as uranyl-carbonate-hydroxide complexes.
- Boron is slowly released from a solid phase to soluble forms through a process called "regeneration".

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Chapter 6
Uptake, Metabolism, and Volatilization of Selenium by Terrestrial Plants

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6.1 Introduction

Selenium (Se) in irrigation drainage water became a major concern after it was implicated in deaths and deformities of waterfowl at Kesterson Reservoir (Presser and Ohlendorf 1987; Ohlendorf 1989). Subsequently, other areas with irrigation-induced Se problems (or potential problems) have been identified in several western U.S. states (Seiler 1995), reinforcing the status of Se as a U.S. Environmental Protection Agency priority pollutant (Keith and Telliard 1979). Selenium hazards are most often associated with semi-arid climates, geologic sources in proximity to large irrigation projects, and hydrologically-closed basins, all of which lead to significant evapoconcentration of dissolved Se (Seiler 1995). Some 17 % of western US land is found on Cretaceous sediments high in Se. When this land is used for irrigated agriculture, Se concentrations in drainage water can reach mg per L levels. Several measures for mitigating these high levels of Se have been proposed, including phytoremediation and phytoextraction of Se-enriched soils and sediments.

Phytoremediation was a newly burgeoning field in the mid-1980s and was being scrutinized as an environmentally benign cleanup technology for a variety of soil contaminants, both inorganic and organic. Researchers affiliated with the UC

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_6,

Salinity Drainage Task Force began evaluating candidate species for deployment in the San Joaquin Valley (SJV) to help reduce soil burdens of Se (Banuelos et al. 1990; Parker et al. 1991; Duckart et al. 1992). The ecotoxicological history of Se actually dates back almost 200 years, to the first reports from South Dakota concerning "alkali" disease in livestock grazing in certain areas. Much later, in the 1920s, high Se levels in forage were found to be the causative agent of alkali disease (Mayland et al. 1989), and this finding spurred numerous investigations into the "geobotany" of Se, culminating in Rosenfeld and Beath's (1964) classic treatise on the subject. Among the most notable findings of this early work was the identification of several unique plant taxa, all endemic to the western USA, capable of accumulating Se to hundreds or even thousands of mg per kg. The latter were originally termed "primary accumulators", while the former were categorized as "secondary accumulators" (Rosenfeld and Beath 1964). The term "hyperaccumulator" was coined much later in reference to plants that accumulate nickel (Ni) to very high concentrations (Brooks et al. 1977) and subsequently extended to a variety of metals and metalloids. Thus, in the mid-1980s, there was actually a large amount of preexisting groundwork concerning the botany and plant physiology of Se accumulation, so that adapting that knowledge base to a phytoremediation context was a logical and facile next step.

Successful phytoremediation using terrestrial plants has always hinged on their tendency to take up Se, predominantly (but not exclusively) as selenate (SeO_4^{2-}), from the soil and accumulate the Se in their shoots (Banuelos et al. 1990, 1997a; Parker et al. 1991). The seleniferous plant material could be harvested and landfilled, used as a supplement in low-Se animal diets, or incorporated back into the soil to promote microbial Se volatilization (Thompson-Eagle et al. 1989). Plant-enhanced volatilization of methyl-selenide compounds from leaves and/or the rhizosphere has also been a topic of considerable interest (Terry et al. 1992; Zayed and Terry 1994), because it can augment Se removal in harvested shoots. The search for superior plant taxa has been an ongoing effort (e.g., Banuelos et al. 1997b; Feist and Parker 2001), as the ideal candidate for Se phytoremediation should be tolerant of the wide range of aerial and edaphic conditions found in the western USA, including heat, drought, soil salinity, and high soil boron (B) (Parker et al. 1991). Intuitively, the ideal phytoremediation candidate should also have a rapid growth rate and large biomass production, along with the ability to accumulate high concentrations of Se in shoot tissue under relevant soil chemical conditions.

The plant taxa utilized in the majority of Se phytoremediation-oriented studies fall into one of three broad categories: Some of the true primary accumulators, all from the *Astragalus* and *Stanley* genera have been examined in a handful of evaluation studies (e.g., Parker et al. 1991, 2003; Bell et al. 1992; Duckart et al. 1992; Retana et al. 1993). A larger number of studies have focused on certain fast-growing *Brassica* species that seem to be secondary Se accumulators, probably because they are avid accumulators of sulfur (S) (Banuelos et al. 1997a; Parker et al. 2003). The third group of studies has utilized genetically modified plants, usually *B. juncea*, in which one or more biochemical/physiological traits is overexpressed in an attempt to boost Se removal rates from soil (e.g., Pilon-Smits et al. 1999; LeDuc et al. 2004).

Our purpose here is to provide an overview of the most salient studies concerning plant uptake, metabolism, and volatilization of Se, framed in the context of phytoremediation approaches to the Se problem in the Central Valley, with an emphasis on field-based validation of the approach.

6.2 Plant Uptake and Translocation of Selenium

6.2.1 Primary Accumulators

Because Se is so chemically similar to sulfur (S), higher plants (and other organisms) tend to take up and metabolize Se readily via S transporters and pathways. Since replacement of S by Se in proteins and other S compounds disrupts the function of these molecules, Se is toxic at elevated levels to most organisms, and the gap between sufficiency and toxicity is often described as narrow (Mayland et al. 1989; McLaughlin et al. 1999). Unlike animals, many bacteria, certain green algae, and higher plants do not seem to require Se, but they nonetheless readily take it up from their environment and incorporate it into organic compounds.

There is a very broad range in the tendency for higher plants to take up and accumulate Se in their aerial parts, both across species (interspecific variation) and within species (intraspecific variation). The primary accumulators of Se (a synonym for hyperaccumulators) are from the *Brassicaceae*, *Fabaceae* and *Asteraceae* families, are endemic to naturally seleniferous soils, and can exhibit shoot Se concentrations as high as 10×10^3 mg kg^{-1} dry weight (1 % of DW) in field-grown specimens; whereas, in Se nonaccumulators, Se concentrations greater than 100 mg kg^{-1} are relatively rare (Rosenfeld and Beath 1964; Parker and Page 1994; Terry et al. 2000). Selenium hyperaccumulators preferentially take up Se over S, and their Se accumulation to percent levels in leaf tissue does not result in toxicity (Neuhierl and Böck 1996; LeDuc et al. 2004). There is evidence that Se hyperaccumulators can distinguish between S and Se (Bell et al. 1992; White et al. 2007) and have Se-specific metabolism, as discussed below. It has been suggested that Se could be an essential element for hyperaccumulators, since hyperaccumulators sometimes exhibit significantly better growth in the presence of Se (Terry et al. 2000), but there is, to date, no compelling evidence that these plants require Se to complete their life cycle. The observed positive growth responses to Se by hyperaccumulators may instead be due to alleviation of phosphorus (P) toxicity in hydroponic culture, since Se accumulation was considerably less pronounced when plants were grown at lower phosphorus levels (Broyer et al. 1972).

6.2.2 Secondary Accumulators

The secondary Se accumulators include a broad array of selected species from the genera *Aster*, *Astragalus*, *Atriplex*, *Castilleja*, *Comandra*, *Grayia*, *Grindelia*, *Gutierrezia*, and *Machaeranthera* that may accumulate Se to levels between 100 and 1,000 mg kg^{-1} (Parker and Page 1994); they have been little studied with respect to the physiology or biochemistry of Se uptake and accumulation. Moreover, plants that have a general tendency to accumulate high levels of S compounds, such as members of the *Brassica* genus (mustards and cabbages), also are good accumulators of Se and can rightfully be grouped with the other secondary accumulators (Terry et al. 2000). These plants include the fast-growing agronomic species *B. juncea*, which has been studied extensively and genetically engineered for purposes of phytoremediation (see later sections). The secondary Se accumulator species probably lack any Se-specific transporters or metabolic pathways, but merely take up and metabolize Se and S indiscriminately, simply at elevated rates compared to nonaccumulators. A similar generalization can be made for the nonaccumulators, but there do seem to be a few notable exceptions (see next section). Among the nonaccumulators, the monocot grasses (*Poaceae*) generally have lower shoot Se levels than do dicotyledonous plants (Rosenfeld and Beath 1964; Parker and Page 1994).

Intraspecific variation in Se accumulation has been studied much less, but Feist and Parker (2001) reported that 16 populations of *Stanleya pinnata* (an identified Se hyperaccumulator), collected from around the western USA, exhibited 1.4- to 3.6-fold differences in shoot Se concentration when grown in a "common garden" greenhouse environment utilizing varying selenate and sulfate levels in sand culture. Moreover, shoot levels were positively correlated with soil Se levels at the site where each ecotype was collected. Crop plants may exhibit genotypic differences in Se accumulation, but there is a paucity of common-garden experiments wherein environmental and/or soil factors can be eliminated as possible sources of variation (see Zhu et al. 2009 for references and a recent review).

The form of soluble Se initially absorbed can profoundly influence its root uptake and subsequent fate. Selenate is the dominant water-soluble form of soil Se, especially under alkaline and aerobic conditions. Although also present in many soils, selenite is generally less bioavailable to plants than is selenate, because the former is more strongly adsorbed. Many nutrient solution studies have utilized selenate (Bell et al. 1992; Feist and Parker 2001), but a few have utilized selenite instead (e.g., Broyer et al. 1972), and some have compared the two inorganic forms (Lewis et al. 1974; Zayed et al. 1998; de Souza et al. 1998; Hopper and Parker 1999). Relatively few studies have examined the uptake of soluble organic forms, such as Se-methionine (Abrams et al. 1990; Zayed et al. 1998). All are "plant available" (Terry et al. 2000), but their comparative bioavailability at equal solution concentrations is confounded by other influential factors, especially the presence of other, "competitive" anions.

Plant uptake of selenate can be inhibited by an assortment of other anions (Khattak et al. 1991), of which sulfate is both the most effective and the most studied (Mikkelsen et al. 1988; Bell et al. 1992; Wu and Huang 1992; Feist and Parker 2001). It is clear that uptake of selenate can be mediated by sulfate transporters, owing to the chemical similarity between selenate and sulfate (Terry et al. 2000). The selectivity of these transporters for selenate and sulfate varies between plant species and with nutritional status (White et al. 2004). It has been suggested that the selectivity of the transport pathway for sulfate over selenate is lower at higher external sulfate concentrations, and that the inducible sulfate transporters have higher selectivity for sulfate over selenate than do the constitutively active sulfate transporters (White et al. 2004). Different sulfate transporters in a single plant may also have somewhat different selectivity for sulfate versus selenate, as had been suggested in studies using *Arabidopsis* (El Kassis et al. 2007; Barberon et al. 2008). Further improvements in our understanding of the selectivity of different sulfate transporters at the molecular level might help the development of plants higher (or lower) in Se via genetic engineering.

The sulfate/selenate transporters in the Se hyperaccumulators have not yet been studied directly, but these taxa are uniquely characterized by generally higher leaf Se concentrations, a higher Se:S ratio, and a higher shoot:root Se concentration ratio (Bell et al. 1992; Feist and Parker 2001; White et al. 2007). These observations are consistent with altered regulation of sulfate/selenate transporters, and/or the presence of specialized Se-specific transporters, perhaps exclusive selenate transporters that have evolved from sulfate transporters. Moreover, a recent study of seasonal fluctuations in Se and S levels in Se hyperaccumulators and related nonaccumulators growing on the same field site suggested independent fluxes of Se and S only in the hyperaccumulators (Galeas et al. 2007). Thus, whole-plant level Se fluxes seem to be rather specialized in Se hyperaccumulators and also distinct from S movement. In the future, it will be particularly instructive to study the properties of the sulfate transporter homologues in Se-hyperaccumulating taxa.

Much less is known about (a) the mechanisms of plant uptake of selenite, which might be more important in neutral-to-acidic soils, especially under reduced soil conditions, and (b) the uptake of organic selenium compounds. Historically, it has been stipulated that selenite uptake by plant roots is not metabolically dependent and does not involve specific ion transporters (Arvy 1993; Terry et al. 2000). But, some more recent results showed that selenite uptake by wheat could be suppressed by a metabolic inhibitor, inhibited by phosphate in the nutrient solution, and enhanced by P deficiency (Li et al. 2008). It was previously argued that inconsistency in the rate of selenite uptake by plants could be ascribed to different phosphate concentrations present in the growth solutions used for different studies (Hopper and Parker 1999). Phosphate is typically present at millimolar levels in almost all nutrient solutions, but at only micromolar levels in soil solutions *in situ* (Parker and Norvell 1999). Increasing P above a basal level of 2 μM in ten-fold increments significantly reduced both Se phytotoxicity and Se uptake in solution-cultured plants supplied with selenite, but the effect was less dramatic than a similar, relative increase in sulfate for selenate-grown plants (Hopper and Parker 1999). Thus, quantitative comparisons of selenite and selenate availability to plants require careful consideration of the in-solution phosphate and sulfate concentration levels, respectively.

Translocation of Se from root to shoot depends on which form (or species) of Se is supplied to the plant. In plants exposed to selenate, Se is readily translocated to the shoot, and selenate is the predominant species found in the xylem sap (Li et al. 2008). In contrast, with selenite-treated plants, most of the Se remains in the roots, and little selenite is detected in the xylem sap. Selenite taken up by roots is readily converted to other forms, including selenomethionine (SeMet) and selenomethionine Se-oxide hydrate (SeOMet), but mostly into unidentified and water-insoluble forms (Li et al. 2008). Thus, Se translocation from root to shoot is generally much lower in plants fed with selenite than those fed with selenate (Arvy 1993; de Souza et al. 1998; Hopper and Parker 1999; Li et al. 2008).

6.3 Selenium Metabolism and Volatilization in Terrestrial Plants

The metabolic fate of Se taken up by plants has been reviewed frequently (e.g., Brown and Shrift 1982; Laüchli 1993; Terry et al. 2000; Sors et al. 2005; Pilon-Smits and Quinn 2010), and a similarly detailed review is beyond the scope of this chapter. Instead, we present a brief overview of Se metabolism in plants, both in non-hyperaccumulators and in hyperaccumulators, which is then linked to the issue of plant-mediated volatilization.

In all plants, typically is inorganic selenate is transported to the leaf chloroplast, where it is reduced first to selenite and then further reduced and assimilated into organic Se (Fig. 6.1). Because of the chemical similarities between Se and S, selenate and selenite are readily assimilated by the S-metabolizing enzymes of the plant, and Se can thus be nonspecifically incorporated into almost any S compound (Terry et al. 2000). The first stable, organic form of Se produced is selenocysteine (SeCys). This amino acid can be incorporated nonspecifically into proteins in lieu of cysteine (Cys), leading to phytotoxicity. An alternative fate of SeCys is ultimate conversion to selenomethionine (SeMet), which also can be incorporated mistakenly into proteins, with generally less harmful effects (Fig. 6.1). The SeMet can also be volatilized, converted to volatile dimethylselenide (DMSe), offering a release valve for excess Se from the plant (Lewis et al. 1966 and see below), while SeCys can also be converted in plants to insoluble, elemental Se plus alanine (Pilon et al. 2003). The elemental Se is probably innocuous, as many bacteria use a similar Se detoxification mechanism. Several lines of evidence (de Souza et al. 1998; Pilon-Smits et al. 1999) suggest that the assimilation of Se from selenate is rate-limited by low levels of ATP sulfurylase (APS), the first enzyme needed for the conversion of selenate to selenite (Fig. 6.1).

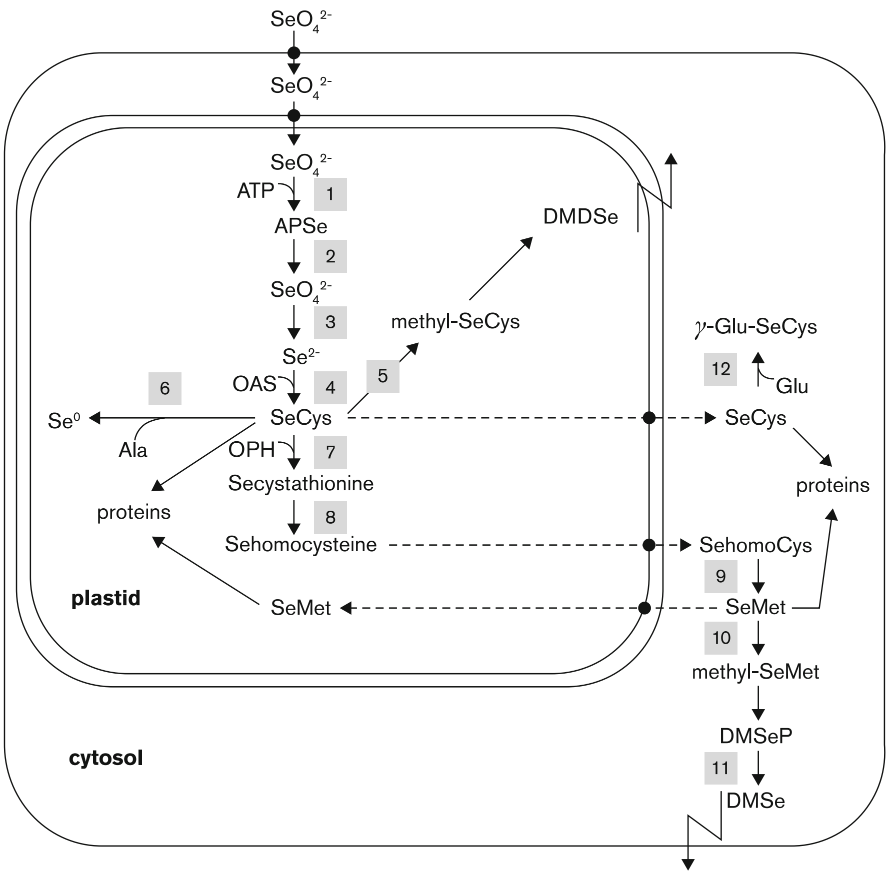

Fig. 6.1 Biochemical pathways of selenium uptake of bacteria, certain green algae, and higher plants

The forms of Se (Se speciation) that tend to accumulate in plant tissues are important from two perspectives: (a) understanding metabolic pathways and (b) assessing the nutritive value of Se-containing foods; they have been investigated in several plant species. Selenium speciation varies both with plant species and the form of Se provided to the plant (e.g., Wang et al. 1996; Grant et al. 2004; Ximenez-Embun et al. 2004; Kapolna et al. 2007). For example, in *Brassica juncea* (Indian mustard), when the plant is fed with selenate, the main Se species is selenate; whereas, in plants fed with selenite, SeMet and SeOMet tend to dominate (Kahakachchi et al. 2004). This pattern likely applies to many, if not most, nonaccumulators, and to secondary Se accumulators, at least with respect to leaf tissues. In cereal crops, the grain Se seems to be dominated by SeMet (~60–80 % of the total Se) (Stadlober et al. 2001).

Beginning in the late 1980s, the increased emphasis on phytoremediation led to a renewed interest in phytovolatilization of methylated Se compounds, which could represent an "added value" to phytoextraction strategies (Banuelos et al. 1997a, b).

The rates of Se volatilization vary greatly among species, with the hyperaccumulator *A. bisulcatus* often exhibiting the highest rates (Duckart et al. 1992; Terry et al. 1992). Dimethyl selenide (DMSe) is the predominant volatile Se compound produced by nonaccumulators and secondary Se accumulators (Lewis et al. 1974). However, a number of rate-limiting steps must be overcome, including the incremental reduction of selenate by APS described above. In addition, de Souza et al. (2000) found that supplying plants with dimethylselenopropionate (DMSeP) results in a remarkable increase in volatilization, and thus proposed that conversion of SeMet to DMSeP is most likely a key rate-limiting step for Se volatilization. Subsequently, overexpression in *B. juncea* of the first enzyme in the conversion of SeCys to SeMet, cystathionine gamma synthase (CγS), resulted in a two- to three-fold higher volatilization rate compared to untransformed plants (Van Huysen et al. 2003).

In addition to these metabolic "bottlenecks" to DMSe production and release to the atmosphere, Pilon-Smits et al. (1999) further suggested there may often be some complex whole-plant cycling involved. They proposed that after selenate is absorbed and translocated to the leaves, it is converted to organic Se. One (or more) of the soluble organic intermediates is then translocated back to the roots via the phloem, where the majority of the volatilization takes place (Zayed and Terry 1994). Alternatively, the organic Se could be incorporated into proteins in place of their sulfur analogues, perhaps leading to phytotoxicity (Terry et al. 2000).

Perhaps unsurprisingly, the quantities of Se volatilized by non-hyperaccumulators depend on the form of Se supplied to the plants, with SeMet $>$ selenite $>$ selenate (Lewis et al. 1966; Zayed et al. 1998). This result is consistent with the metabolic steps (Fig. 6.1) and rate limitations in the production of DMSe. The observation that increasing sulfate levels decreases volatilization in selenate-supplied plants, but not in those exposed to selenite or SeMet (Zayed and Terry 1994; Zayed et al. 1998), is also consistent with the concept that the most profound effect of sulfate is to inhibit the initial entry of selenate into the plant.

In certain plant taxa, notably the Se hyperaccumulators, SeCys can be methylated to form Se-methylselenocysteine (SeMeCys) by the enzyme selenocysteine methyltransferase (SMT) (Neuhierl and Böck 1996). This form of Se was first identified in the early 1960s (Trelease et al. 1960; Shrift and Virupaksha 1963), and it has long been believed that this form of Se can safely be accumulated to high concentrations, since it is not incorporated into proteins; SeMeCys likely plays a central role in the detoxification of Se in the hyperaccumulators (Neuhierl et al. 1999). Interestingly, SeMeCys is also a major Se compound found in Se-enriched garlic (*Allium sativum*), onion (*A. cepa*), leek (*A. ampeloprasum*) and broccoli (*Brassica oleracea*), accounting for approximately half of the total Se (Whanger 2002). Recent studies with the hyperaccumulator *Stanleya pinnata* have also found that the soluble Se in the shoots is dominated by amino acid forms (Zhang and Frankenberger 2001). In addition to SeMeCys, the Se hyperaccumulators may also store Se in the leaves as γ-glutamyl-Se-methylselenocysteine, and as selenocystathione (Terry et al. 2000).

While much is known about Se volatilization in secondary Se accumulators, such as *B. juncea*, comparatively little is known about volatilization from the true hyperaccumulators. An early study of *Astragalus racemosus* collected volatile Se compounds on active carbon and found four compounds, of which only one could be identified – dimethyldiselenide (DMDSe) (Evans et al. 1968). This seminal paper led to repeated speculation that Se hyperaccumulators follow a distinctly different metabolic pathway, wherein DMDSe is produced directly from SeMeCys (e.g., Brown and Shrift 1982; Terry et al. 2000; Sors et al. 2005). Only recently has further credence been given to the significance of DMeDSe. Using sophisticated gas headspace analysis, Kubachka et al. (2007) were able to show the increased significance of DMeDSe (relative to DMeSe) in transgenic *B. juncea* in which the overexpression of SMT led to enhanced accumulation of SeMeCys relative to the wild type. Thus, a *B. juncea* genotype seems to share many of the traits of the true hyperaccumulators (LeDuc et al. 2006). The literature often describes Se hyperaccumulating species as having a characteristic odor, which *could* be a volatile Se compound, but this remains speculation at present.

To date, studies of plant Se metabolism, including volatilization, have generally been done using nonsterile culture methods. Since all plants live in association with a diverse array of bacteria and fungi, and many microbes can also metabolize and volatilize Se, plant-associated microbes may play a key role in plant Se accumulation, and especially in Se volatilization. For example, in broccoli (*B. oleracea*) 95 % of the Se volatilized by roots was inhibited when roots were treated with antibiotics (Zayed and Terry 1994). Similarly, *B. juncea* plants treated with the antibiotic ampicillin volatilized 30 % less Se, and even accumulated 70 % less Se than the control plants. In addition, *B. juncea* plants grown from surface-sterilized seeds that were subsequently inoculated with rhizospheric bacteria accumulated five-fold more Se and volatilized four-fold more Se than uninoculated controls (Zayed and Terry 1994). The mechanisms for the stimulatory effect of the bacteria appeared to be both an enhancement of root growth, as well as direct stimulation of Se/S uptake and assimilation. Plants inoculated with rhizospheric bacteria had an increased root surface area and the culture media contained nine-fold higher serine levels than control plants. O-acetylserine (OAS) is known to stimulate S uptake and assimilation (de Souza et al. 1999).

While there is convincing evidence that bacteria mediate plant uptake and volatilization of Se, much less is known about a possible role for plant-associated fungi. In one study, the nonaccumulator ryegrass accumulated less Se when inoculated with the mycorrhizal fungus *Glomus mosseae* in comparison to controls lacking the fungal symbiont (Munier-Lamy et al. 2007). Moreover, virtually nothing is known about the role of endophytic microbes in Se uptake and volatilization. These will be interesting areas for further study.

6.4 Genetic Engineering for Enhanced Phytoremediation

As discussed previously, all plants can take up inorganic selenate and selenite and further assimilate them to SeCys and other organic compounds, including some volatile forms. Hyperaccumulators of Se may have additional metabolic pathways for Se, notably the methylation of SeCys, and most likely the conversion of MeSeCys to volatile DMDSe. To further enhance plant selenium accumulation, tolerance, and volatilization, various transgenic approaches have been used, primarily in *B. juncea*.

The main thrust of the transgenic work has involved upregulation of key genes involved in S/Se assimilation and volatilization. As previously mentioned, overexpression of the gene for the first enzyme in the reduction of selenate to selenite conversion, ATP sulfurylase (APS), resulted in enhanced selenate reduction, as evidenced by the increase in organic forms of Se when supplied with selenate, while wildtype controls accumulated selenate (Pilon-Smits et al. 1999). The APS transgenics accumulated two- to three-fold more Se than the wild type controls, and 1.5-fold more S. The APS plants also tolerated the accumulated Se better than the wild type controls, perhaps because of the different form of Se accumulated, but the Se volatilization rate was not enhanced in the APS transgenics.

Van Huysen et al. (2003) subsequently showed that overexpression of the first enzyme in the conversion of SeCys to SeMet, cystathionine gamma synthase (CgS), resulted in substantial increases (two- or three-fold) in volatilization rates from *B. juncea*. Probably as a result of their enhanced volatilization, the CgS transgenics accumulated 40 % less Se in their tissues than the wildtypes, and were also more Se-tolerant, probably due to their lower tissue Se levels. Another transgenic approach targeted the nonspecific incorporation of SeCys into proteins by overexpressing a mouse selenocysteine lyase (SL), an enzyme that specifically breaks down SeCys into alanine and elemental Se, in both *A. thaliana* and *B. juncea* (Pilon et al. 2003; Garifullina et al. 2003). The SL transgenics showed reduced Se incorporation into proteins, as well as enhanced Se accumulation (up to two-fold) compared to wildtype plants.

Another approach to enhancing Se tolerance was to overexpress SeCys methyltransferase (SMT) from the Se hyperaccumulator *A. bisulcatus* in both *A. thaliana* and *B. juncea* (Ellis et al. 2004; LeDuc et al. 2004). The SMT transgenics showed enhanced Se accumulation in the form of methyl-SeCys, as well as enhanced Se tolerance, and greater production of volatile Se, most likely as DMDSe (Kubachka et al. 2007). While the expression of SMT enhanced Se tolerance, accumulation, and volatilization, the effects were more pronounced when the plants were supplied with selenite, as opposed to selenate, consistent with the perspective that conversion of selenate to selenite was the rate-limiting step for the ultimate production of MeSeCys. To overcome this rate limitation, APS and SMT transgenics were crossed to create double-transgenic plants that overexpress both APS and SMT. The APS × SMT double transgenics accumulated up to nine-fold more Se than the wildtypes, and most of the increase could be ascribed to accumulation of the nontoxic MeSeCys (LeDuc et al. 2006). The APSxSMT plants accumulated almost twice as much MeSeCys as the single SMT transgenics, but Se tolerance was similar in the single and double transgenics.

These various transgenics have been tested further to validate their potential beyond the laboratory, where Se accumulation was up to nine-fold higher and volatilization rates were up to three-fold greater. When grown in a naturally seleniferous soil in greenhouse pots, the APS transgenics accumulated Se to three-fold higher levels than wildtype *B. juncea*, while the CgS transgenics contained 40 % lower Se levels than the wildtypes (Van Huysen et al. 2004), in agreement with the laboratory results. Plant growth was the same for all plant types in this experiment. Subsequently, field experiments were carried out on a Se-contaminated sediment in the San Joaquin Valley (Banuelos et al. 2005a, b, 2007), and again results were obtained that agreed with the earlier laboratory experiments (see last section for details). Thus, the results obtained with the transgenics in the greenhouse or in the field are similar to those obtained under controlled laboratory conditions, and the enhanced Se accumulation, volatilization and/or tolerance exhibited by the transgenics are promising traits for use in phytoremediation.

6.5 Other Desirable Plant Traits for Se Phytoremediation

Early on, researchers working as part of the UC Salinity/Drainage Task Force recognized that, if terrestrial plants were to be used to help dissipate Se from high-Se soils or sediments, they would need to be tolerant of an array of adverse aerial and edaphic conditions associated with the relevant semiarid sites in California and elsewhere (Parker and Page 1994). Foremost was concern about high soil levels of both salinity and boron (B), as the substrates in which Se was the highest (such as Kesterson) were also laden with salts. Moreover, plants were needed that could also tolerate heat and drought, and that had agronomic traits useful at the field scale. Initially, these requirements presented something of a fundamental conundrum: the Se hyperaccumulators (e.g., some *Astragalus* and *Stanleya* species) had never been improved via plant breeding, although they are broadly adapted to some rather harsh environments in the western USA Other candidates included improved crop cultivars of species such as *B. juncea*, but these had not been bred specifically for highly saline soils.

In an early study, Parker et al. (1991) used greenhouse sand culture to screen a number of genotypes of the genera *Astragalus*, *Leucaena*, *Medicago*, *Trifolium*, *Elymus*, *Elytrigia*, *Festuca*, *Leymus*, *Oryzopsis*, *Psathyrostachys*, *Puccinellia*, and *Sporobolus* for tolerance to salinity and B. Salinity treatments were specifically designed to mimic the sulfate-rich drainage waters found within the Westside of the San Joaquin Valley. Considerable variation in tolerance to salinity, both within and across species, was observed during seed germination, but B concentrations up to 4.0 mM had little effect. The most promising genotypes, representing some 15 species, were then tested for salinity and B tolerance during the seedling growth stage. Lines of five species (two Se-hyperaccumulators, *Astragalus bisulcatus* and *A. racemosus*, and three grasses, *Elytrigia pontica*, *Puccinellia distans*, and *Sporobolus airoides*) appeared the most promising, as they grew well up to ~15 dS m^{-1} salinity, and were again unaffected by B at 4.0 mM.

Retana et al. (1993) then attempted to rear four of these salt- and B-tolerant genotypes (along with Indian ricegrass [*Oryzopsis hymenoides*]) in a greenhouse column study using a soil from Kesterson to assess both growth and uptake of various trace elements (As, B, Mo, Se, U, V). Soil columns were reconstructed to reflect the original depth profile in the field, but subjected to a pre-planting leaching treatment to reduce salinity in the seed zone. All five genotypes were established successfully, with the alkali sacaton and tall wheatgrass yielding the greatest biomass; however, the shoot Se levels were comparatively low in these two grasses. *Astragalus bisulcatus* and *A. racemosus* were much slower-growing, but were able to persist on very saline soil, accumulating by far the greatest amount of Se in their aerial tissues (Retana et al. 1993).

Banuelos et al. (1990) evaluated the effects of Se, salinity, and B on plant growth and elemental accumulation in solution-cultured *B. juncea*. Salinity (chloride-dominated) up to 15 dS m^{-1} and B up to 1.4 mM both seemed to reduce shoot yield, but by less than 40 %, even in concert. In a follow-up study, Banuelos et al. (1996) screened multiple genotypes of *B. napus* (canola), *Hibiscus cannibinus* (kenaf), *Festuca arundinacea* (tall fescue), and *Lotus tenuis* (birdsfoot trefoil) for salinity tolerance (again, chloride-based, up to 20 dS m^{-1} in soil pots), in conjunction with Se uptake. Overall, canola exhibited the best combination of salinity tolerance, biomass production, and Se accumulation. Wu and Huang (1991) also examined 13 tall fescue lines from around the world and found a correlation between salt tolerance and Se tolerance, although this species is not particularly salt-tolerant, generally. Subsequently, Parker et al. (2003) did a comparative study of *B. juncea* and the hyperaccumulator *S. pinnata* using the same greenhouse methods employed earlier (Parker et al. 1991). Neither species was particularly tolerant of salinity (as compared to alkali sacaton which was grown alongside), although *S. pinnata* exhibited the desired tolerance to excess B. That *S. pinnata* exhibits ecotypic variation in Se accumulation, as well as broad geographical distribution (Feist and Parker 2001), gave rise to some optimism that more salt-tolerant germplasm may exist (Parker et al. 2003). Moreover, *S. pinnata* is a perennial that responded favorably to repeated cutting in the greenhouse, even when subjected to salinity or B stress, a trait that could prove valuable in field-scale phytoremediation. In contrast, *B. juncea* exhibited much poorer survival and regrowth under high salinity and B.

Relatively few other studies have been conducted to screen diverse plant taxa for tolerance to salinity or B, and we know of none that have specifically addressed other traits, such as heat or drought tolerance. There are, however, some field trials from which the ability to withstand these conditions can be inferred, and these are summarized briefly below.

6.6 Phytomanagement of Se Under Field Conditions

After more than two decades of extensive research supported by the UC Salinity/Drainage Program, many strategies have been suggested to manage soil Se levels. Removal of Se from soil with conventional physical and chemical techniques is prohibitively expensive, as are excavation and burial. Thus, much of the emphasis has been upon the “green” technology broadly known as phytoremediation (Pilon-Smits 2005). In its broadest sense, the strategy uses plants to manage contaminated water and/or soil via accumulation (extraction), enhanced volatilization, soil stabilization, rhizofiltration, and degradation/transformation; the first two have been the foci of most of the relevant Se studies. Effective phytoextraction at the field scale requires that plants produce relatively large amounts of biomass, absorb soil Se and translocate it such that it can be harvested and removed from the site, and/or promote volatilization of nontoxic gases (e.g., dimethyl selenide, DMSe).

Phytomanagement of Se both with enhanced phytoextraction and/or volatilization goals requires cogent use of the most appropriate plant species. Plants with differing properties may be needed to obtain an effective overall technology, especially under field conditions, and may involve the use of hyperaccumulator plants, agronomically improved crops, particular plant cultivars, or even genetically-modified plants (Terry et al. 2000; Banuelos et al. 2007). Although plant selection is critical for successful phytoremediation, understanding the following factors is also essential when considering phytomanagement strategies under field conditions:

1. Remediation site: soil vs. water; adverse growing conditions, presence of shallow water tables, field variability.
2. Agronomics, e.g., cultivation, management practices, crop rotation, including availability of water for irrigation.
3. Crop disposal or utilization of harvested material, economic sustainability.
4. Time frame: how long-term is the process?
5. Parameters used to measure success: fraction removed, fraction converted to less toxic forms.
6. Grower acceptance of alternative use crops

It is beyond the scope of this chapter to review all of the published field studies of Se phytomanagement. Instead, we review key examples that help illustrate the foregoing, while also demonstrating the application of the more fundamental knowledge gain in the laboratory and greenhouse, which we have reviewed previously.

In an early study, Banuelos et al. (1993) examined the ability of *B. juncea* to dissipate soil Se in a field being irrigated with saline drainage water (Se = 154 μg L^{-1}). Although Se was taken up by the pant and volatilized, the quantities removed were small, and, in fact, insufficient to offset the annual Se inputs from the irrigation water. Similar results were later obtained by Lin et al. (2002). Even with the “best” volatilizer of Se (pickleweed [*Salicornia bigelovii*]), only about 7 % of the annual total Se input was volatilized.

Van Mantgem et al. (1996) studied the ability of two forage species to dissipate Se from Kesterson Reservoir sediments, using “clean” irrigation water. The plants removed a measurable fraction of the more labile forms of Se in the sediment, but these comprised only 10 % of the total Se inventory; there was thus no measurable reduction in total Se in a year-long time frame.

More recently, Banuelos et al. (2005a) looked at phytovolatilization of Se from the heavily contaminated sediments in the San Luis Drain, using several salt-tolerant species and/or genotypes. Volatilization rates were generally low

(<40 μg m^{-2} day^{-2}), which the authors attributed to the high sulfate levels present and to competitive inhibition of uptake and/or metabolism. They concluded that the volatilization process required enhancement in order to be effective. In a follow-up study, amendment of the same sediments with casein or methionine increased volatilization by up to 17-fold (Banuelos and Lin 2007), but with greatly increased operating costs.

Some of the transgenic lines of *B. juncea* described previously have now been tested under field conditions in the San Joaquin Valley. Banuelos et al. (2005b) reported that APS transgenics accumulated Se to four-fold higher levels than did the wildtypes, which is similar to the earlier laboratory and greenhouse results. In a second field experiment on the same high-Se sediment, the SL and SMT transgenics showed two-fold greater Se accumulation relative to the wildtype (Banuelos et al. 2007), also in agreement with earlier laboratory experiments. In both field experiments, biomass was comparable for the different genotypes. Thus, the results obtained with the different transgenics using Se-enriched soils in both greenhouse or field are similar to those obtained under controlled laboratory conditions. The various transgenics showed enhanced Se accumulation, volatilization and/or tolerance, all promising traits for use in phytomanagement as in phytoremediation.

In closing, it is apparent that, under field conditions, phytomanagement of Se will require time (months/years) for effective removal of soluble Se from the soil. The efficacy of phytomanagement is generally greater under controlled greenhouse or microplot conditions, especially if Se is added as soluble selenate. Under field conditions, naturally-occurring soil Se usually exists as a complex melange of forms (e.g., elemental Se, sorbed selenite, organic Se) that are not immediately available for plant uptake and/or for volatilization. When biogeochemical changes occur within the soil profile, which may be achieved partially through growing crops, the bioavailability of Se may be enhanced. Moreover, irrigation cycles (wetting and drying) may enhance the release of Se, such that uptake and/or volatilization are also enhanced. However, lowering the overall concentration of total soil Se with plant-induced and associated microbiological processes will always take time. Under some circumstances, the best we can do under field conditions is to phytomanage soluble Se by promoting both plant uptake and biological volatilization. Unless plants take up Se faster than evapotranspiration occurs, the net effect of irrigating land with any Se-enriched water may be a gradual increase in the soil Se level (Parker and Page 1994), unless dissipation processes can be greatly accelerated.

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Chapter 7
Bacterial Reduction of Selenium

Yiqiang Zhang and William T. Frankenberger, Jr.

7.1 Introduction

Crop irrigation in the western San Joaquin Valley has produced high salt content agricultural drainage water containing selenium (Se) at levels that have ranged from 140 to 1,400 μg L^{-1} (Sylvester 1990), mostly as selenate [Se(VI)], which has created serious hazards to waterfowl foraging and nesting around the wetlands (Presser and Ohlendorf 1987; Ohlendorf 1989). Removing Se from agricultural drainage water before it reaches wetlands would protect wetland wildlife.

The concentration and chemical speciation of Se in terrestrial and aquatic systems are dependent on the governing chemical factors, such as pH, redox potential, solubility and ability to complex with ligands. Microbial metabolism plays a vital role in reactions of Se species. Volatilization through biomethylation is a protective mechanism used by microbes to detoxify their surrounding environment. Our early efforts were focused on isolating and identifying Se-methylating microbes from Se-enriched sites in the San Joaquin Valley and augmenting the formation of volatile organic Se species, such as dimethylselenium (DMSe), thus reducing the potential of Se to be a toxin in terrestrial and aquatic ecosystems. The work was summarized in several technical publications (Thompson-Eagle et al. 1989; Frankenberger and Karlson 1989a, b, 1994a; Frankenberger and Arshad 2001) and resulted in the Frankenberger – Karlson Process (Frankenberger and Karlson 1989a, b, 1994b) patented by U.S. Bureau of Reclamation (Fig. 7.1). In this chapter, we concentrate on the Se bioreductive processes that reduce bioavailable and soluble Se(VI) and Se(IV) to Se(0) precipitate.

In aquatic systems, Se may exist in four oxidation states, −II, 0, IV, and VI. Because elemental Se [Se(0)] exists as an insoluble precipitate, reduction of Se (VI) to Se(0) is a potential method for bioremediation. Bacteria species, such as

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_7,

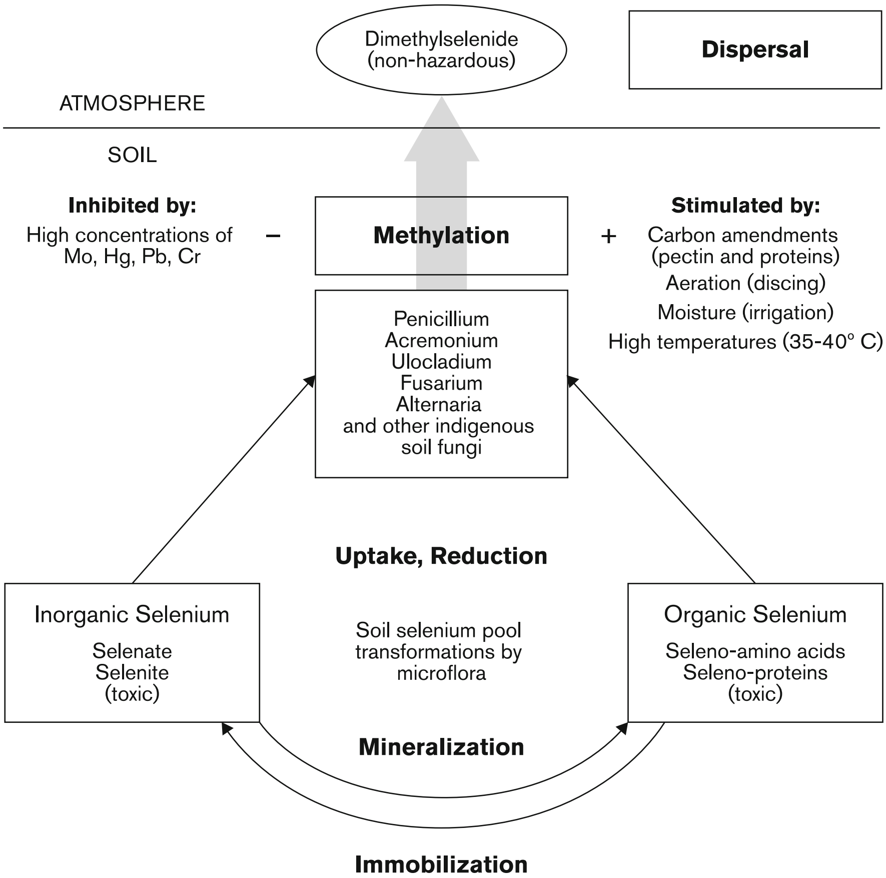

Fig. 7.1 Se volatilization in soils as depicted in the Frankenberger – Karlson selenium detoxification process, U. S. Patent 4,861,482, 1989 (Frankenberger and Karlson 1989a)

Bacillus sp. SF-1, *Bacillus* sp. RS1, *Citerobacter freundii*, *C. braakii*, *Enterobacter cloacae*, *E. taylorae*, *Pseudomonas stutzeri*, *Sulfurospirillum barnesii*, *Thauera selenatis*, and *Wolinella succinogenes*, are capable of reducing Se(VI) to Se (0) (Cantafio et al. 1996; Fujita et al. 1997; Francisco et al. 1992; Lortie et al. 1992; Losi and Frankenberger 1997; Oremland et al. 1999; Zahir et al. 2003; Zhang et al. 2004; Zhang and Frankenberger 2006, 2007). Studies show that these bacteria use Se(VI) as a terminal electron acceptor for anaerobic respiration and use organic carbon sources (yeast extract, acetate, lactate and glucose) as electron donors.

Bacterial reduction of Se(VI) to Se(0) involves a two-step pathway, such that Se (VI) is reduced first to selenite Se(IV) and then Se(IV) is reduced to Se(0). In the process, all three species of Se may be present in the water simultaneously. For effective removal of Se from agricultural drainage water, both Se(VI) and Se(IV)

Table 7.1 Se-reducing bacterial strains isolated from various environmental settings

Bacterium strain	Environment	Reference
Enterobacter taylorae	Rice straw	Zahir et al. (2003)
Pantoea sp. SSS2	Salton Sea sediment, CA	Zhang et al. (2007a)
Citerobacter freundii	Stewart Lake sediment, UT	Zhang et al. (2004)
Klebsiella sp. WRS2	White River sediment, CA	Zhang et al. (2007a)
Shigella sp. WR2	Drainage water from a rice straw treatment system	Zhang et al. (2007a)
Bacillus sp. RS1	Rice straw	Zhang and Frankenberger (2007)
Citerobacter braakii	Drainage water from a rice straw treatment system	Zhang and Frankenberger (2006)

need to be effectively and rapidly reduced. Our laboratory has isolated several Se(VI)-reducing bacteria from different environments (Table 7.1) and used them to reduce Se(VI) to Se(0) in drainage water.

7.2 Factors Affecting Se(VI) Reduction

The amount of salts, competitive electron acceptors, and electron donors present in agricultural drainage (AD) water are three of the important factors affecting Se (VI) reduction. Each liter of the synthetic AD water prepared for the laboratory study contained the following constituents: 1.48 g Na_2SO_4, 0.659 g NaCl, 0.275 g $NaHCO_3$, 0.733 g $CaCl_2.2H_2O$, 0.745 g $MgSO_4$, 0.073 g $(NH_4)_2SO_4$, 0.176 g $Na_2B_4O_7{\cdot}4H_2O$, 0.019 g KCl, 0.044 g $FeCl_2$, 0.0002 g NaH_2PO_4, 0.5 g yeast extract, 0.5 g glucose, and 1 mL trace element solution (Focht 1994). The electrical conductivity (EC) and pH of the final solution were 5.1 dS m^{-1} and 8, respectively. The synthetic AD water was autoclaved and 2,000 μg Se(VI) L^{-1} was added prior to use (Zhang et al. 2003).

7.2.1 Salts

Dissolved salts are unavoidable byproducts of tile drainage from irrigated farmlands in semiarid and arid climates (Oswald et al. 1989). The amounts of dissolved salts in AD water affect the efficiency of reduction of Se(VI) to Se(0) by *E. taylorae* (Fig. 7.2). Reduction of the added Se(VI) proceeded rapidly, with 98 % removal of Se(VI) and Se(IV) during a 7-day period. The efficiency of Se (VI) reduction decreased as the concentration of dissolved salts increased in the synthetic AD water, which was adjusted by adding Na_2SO_4 and NaCl with a weight ratio of 3:1 to an EC range of 25, 50 and 75 dS m^{-1}. The removal of Se(VI) and Se

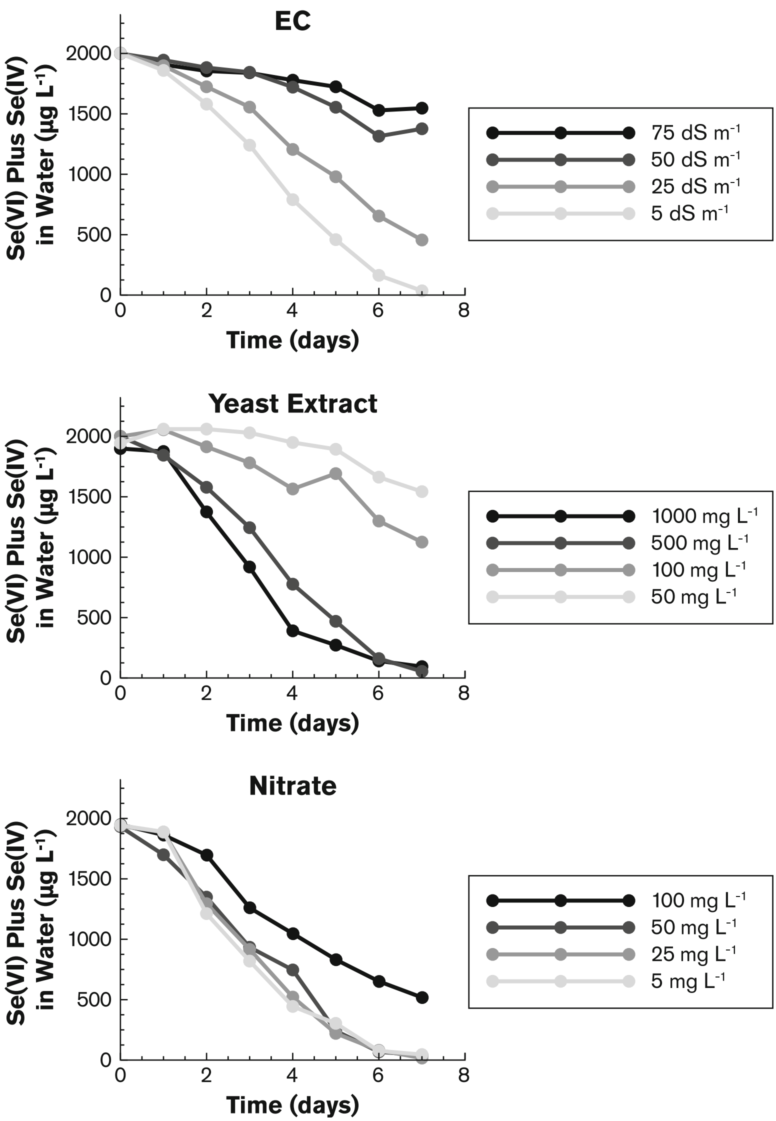

Fig. 7.2 Effects of EC, yeast extract and NO_3^- on reduction of Se(VI) and Se(IV) in synthetic drainage water by *E. taylorae*

(IV) dropped from 75 to 35 %, and finally to 20 %, respectively, in the drainage water having ECs of 25, 50 and 75 dS m^{-1}, respectively. In a study on the effect of SO_4^{2-}, a major anion in AD water, on the rate of Se(VI) reduction, using washed cell suspensions of *D. desulfuricans*, Zehr and Oremland (1987) reported that the rate of Se(VI) reduction was negatively correlated with SO_4^{2-}.

7.2.2 Organic Carbon

In microbial metabolism, organic compounds are used as electron donors and as energy sources for Se bio-reduction. Yeast extract is commonly used in culture media for microbial reduction of Se(VI) to Se(0) (Fujita et al. 1997; Ike et al. 2000; Losi and Frankenberger 1997; Steinberg at al. 1992). Yeast extract is an essential organic carbon source for *E. taylorae* to reduce Se(VI) to Se(0) (Fig. 7.2). In 7 days, 96–98 % of Se(VI) and Se(IV) was removed from the synthetic AD water treated with 0.05–0.1 % yeast extract. In the drainage water treated with 0.005–0.01 % yeast extract, only 20–40 % of Se(VI) and Se(IV) was reduced to Se(0). Fujita et al. (1997) reported that yeast extract, as a growth factor, was required for *Bacillus* sp. SF-1 to reduce Se(VI) to Se(0). Casamino acids, a DNA/RNA mixture, and a vitamin mixture were not able to replace yeast extract.

7.2.3 Electron Acceptor Competition

Nitrate (NO_3) is a common anion found in AD water in the SJV due to application of nitrogen fertilizers on croplands. Nitrate, as a competitive electron acceptor to Se (VI), can negatively affect Se(VI) reduction to Se(0) in aquatic systems, depending on concentration and other factors (Masscheleyn and Patrick 1993; Stolz and Oremland 1999). For example, when the drainage water had a relatively low NO_3^- level of 5–50 mg L^{-1} did not affect the removal of Se(VI) and Se(IV) by *E. taylorae* (Fig. 7.2). Under these conditions, more than 97 % of Se(IV) and Se (IV) was removed via reduction to Se(0) during the 7-day experimental period. However, the removal of Se(VI) and Se(IV) declined to 75 % of the total when the NO_3^- concentration increased to 100 mg L^{-1}. Steinberg et al. (1992) reported that NO_3^- reduction preceded the Se(VI) reduction in an anaerobic freshwater enriched with equal concentrations (20 mM) of Se(VI) and NO_3^-.

7.2.4 Role of Molasses

Selecting an inexpensive and effective organic carbon source was the key to reducing the cost of Se remediation in AD water. Molasses contains relatively high amounts of sucrose, and small amounts of dextrose, fructose, glucose, and some trace metals, such as Fe, Cu, Mn, and Zn (USSC 2003). Due to its low cost, molasses has been used in an experimental algal-bacterial treatment system to support biological removal of Se from AD water (Quinn et al. 2000). In order to find effective Se(VI)-reducing bacteria capable of using molasses to reduce Se (VI) in AD water, we isolated several Se(VI)-reducing bacteria from different environments and screened them, based on their ability to use molasses as an organic carbon source to reduce Se(VI) in drainage water. Each liter of the synthetic AD water used in this study contained the following constituents: 5.918 g Na_2SO_4, 0.989 g NaCl, 0.917 g $CaCl_2.2H_2O$, 1.238 g $MgSO_4$, 0.138 g $NaHCO_3$, 0.091 g K_2HPO_4, 0.027 g $NaNO_3$, 0.0004 g $FeCl_2.4H_2O$, and 1 mL trace element solution (Focht 1994). The drainage water was autoclaved and 1,000 μg L^{-1} of Se(VI) was added prior to use (Zhang et al. 2007a).

Efficiency of Se removal differed in the 0.1 % molasses-added drainage water containing different strains of bacteria (Fig. 7.3). In the aseptic control, there was little change in the Se(VI) concentration in the drainage water during the 11-day experimental period. The Se (VI) concentration decreased in the drainage water in the presence of Se(VI)-reducing bacteria cells. However, the extent of Se (VI) reduction differed in the drainage water inoculated with different bacterial strains. The Se(VI) concentration decreased from 1,000 to 830 μg L^{-1} and to 254 μg L^{-1} after 11 days of incubation in the drainage water inoculated with *Shigella* sp. and *Klebsiella* sp., respectively. Similar reduction of Se(VI) was found in the drainage water in which *C. freundii* was added. Sharp reductions in Se(VI) from 1,000 to 2.99 μg L^{-1} and to 14.02 μg L^{-1} were observed when the drainage water was inoculated with *Pantoea* sp. or *E. taylorae*, respectively. At the end of the experiment, Se(IV), Se(0), and organic Se concentrations were 192, 804, and 1.23 μg L^{-1} and 8.44, 970, and 7.9 μg L^{-1} in the *Pantoea* sp.- and *E. taylorae*-treated drainage water, respectively. Although the *Pantoea* sp. was capable of using molasses effectively to rapidly reduce Se(VI) to Se(IV) as did *E. taylorae*, the *Pantoea* sp. did not reduce Se(IV) to Se(0) as effectively as *E. taylorae*, which emerged as the best bacteria strain among the five tested in using molasses to reduce Se(VI) to Se(0).

Enterobacter taylorae not only demonstrated its ability to use molasses in reducing Se(VI) in the synthetic AD water but also effectively reduced Se in natural AD water that had an EC of 10.3 dS m^{-1} and pH of 7.3 (Fig. 7.4). Natural AD water collected from the western SJV contained 230.9 μg L^{-1} soluble Se, consisting of 218.8 μg L^{-1} Se(VI) plus 12.1 μg L^{-1} Se(IV), and 23 mg L^{-1} of NO_3^--N. By simply adding 0.1 % molasses, there was little change in Se concentration in the natural drainage water for 7 days, until it was inoculated with *E. taylorae*. Then, after inoculation, the Se(VI) decreased rapidly from 218.8 to 15.7 μg L^{-1} in the natural AD water.

7.3 Enhancing Bacterial Reduction of Se(VI) to Se(0)

In the previous sections, we have identified the Se(VI)-reducing bacteria, which are capable of using molasses to reduce Se(VI) to Se(0). In order to increase the efficiency of reducing Se(VI) in AD water, a series of experiments have been conducted.

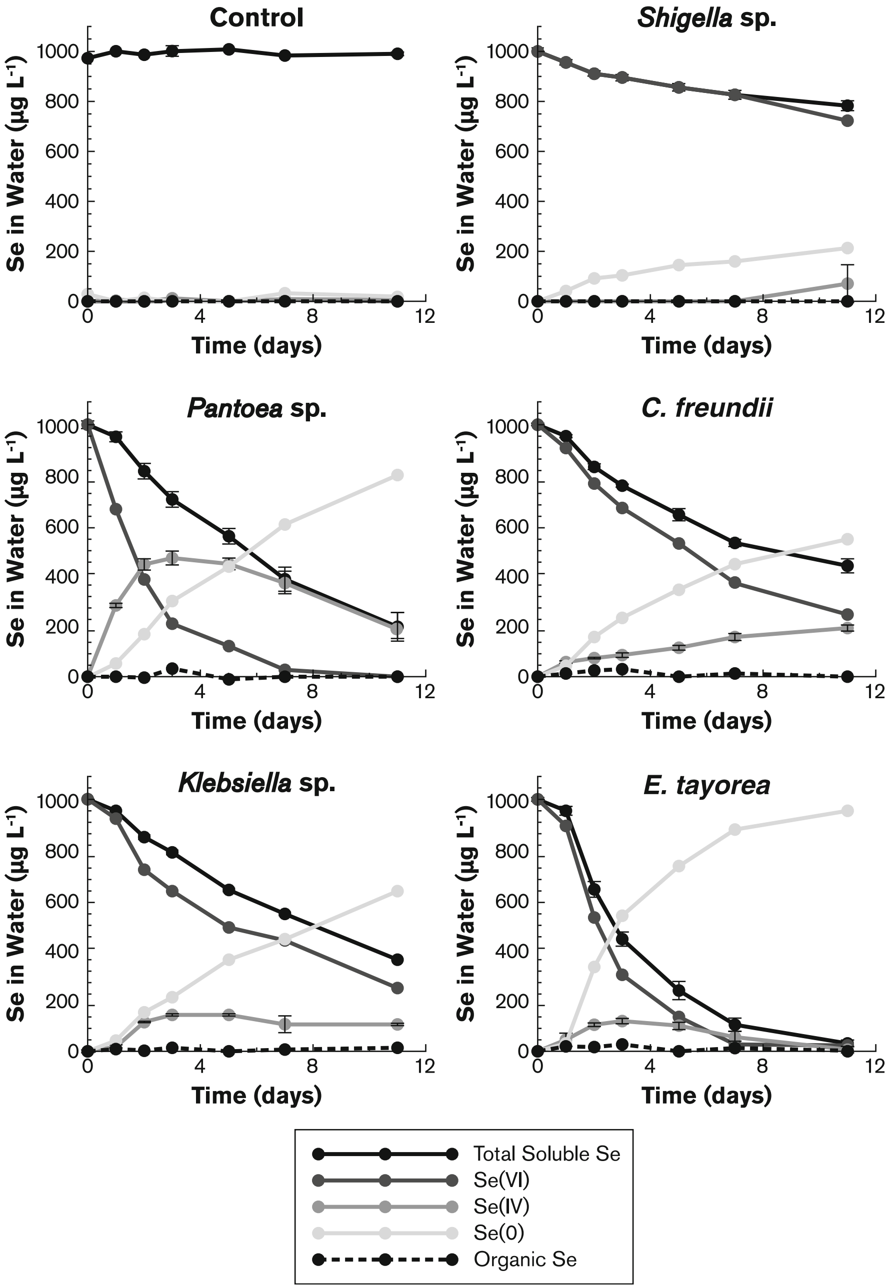

Fig. 7.3 Reduction of Se(VI) to Se(0) in molasses drainage water inoculated with different strain of Se reducing bacteria. Error bar denotes standard deviation of three observations

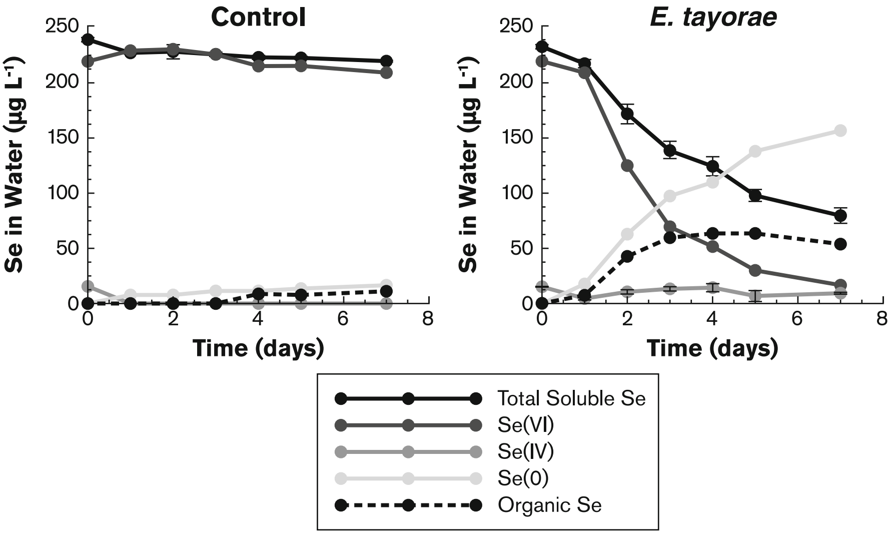

Fig. 7.4 Se removal by *E. taylorae* in 0.1 % molasses treated natural drainage water. Error denotes standard deviation of three observations

7.3.1 Efficacy of Bacillus *sp. RS1 and* Dechloromonas *sp. HZ for Se(VI) Reduction*

In the SJV, AD water typically contains 3–234 mg NO_3^- L^{-1}, with an average of 97 mg NO_3^- L^{-1}, many orders of magnitude higher than the Se concentration of 140–1,400 µg Se L^{-1} in the same waters (Oswald et al. 1989; Cantafio et al. 1996; Sylvester 1990). The redox potential of NO_3^-/N_2 in an aquatic system is very similar to that of Se (VI)/Se(IV) and higher than Se(IV)/Se(0) (Masscheleyn and Patrick 1993); thus NO_3^- may be a competitive electron acceptor affecting Se(VI) reduction to Se(IV) and inhibiting Se(IV) reduction to Se(0) (Fujita et al. 1997; Masscheleyn and Patrick 1993; Steinberg et al. 1992; Weres et al. 1990). The addition of a NO_3^- reducing bacterial strain to the AD water would aid the bacterial reduction of Se(VI) by lowering the NO_3^- concentration in the water, effectively reducing competition. The synthetic AD water used in this study contained Na_2SO_4, 5.921 g; NaCl, 0.989 g; $NaHCO_3$, 0.138 g; $CaCl_2.2H_2O$, 0.917 g; $MgSO_4$, 1.238 g; $Na_2B_4O_7.10H_2O$, 0.529 g; $NaNO_3$, 0.069 g; K_2HPO_4, 0.091 g; $FeCl_2$, 0.0002 g; and 1 mL trace element solution in each liter of solution (Focht 1994). The drainage water containing 0.1 % molasses and Se(VI) was autoclaved prior to use (Zhang et al. 2007).

The NO_3^- concentration affected the Se(VI) reduction in the AD water inoculated by *Bacillus* sp. RS1 and a nitrate reducer, *Dechloromonas* sp. HZ (Fig. 7.5). In drainage water free of NO_3^-, Se(VI) was almost entirely reduced and the resulting Se(0) and Se(IV) accounted for 36 and 63 % of the added Se (1,000 µg L^{-1}) in the drainage water, respectively, during 8 days of incubation.

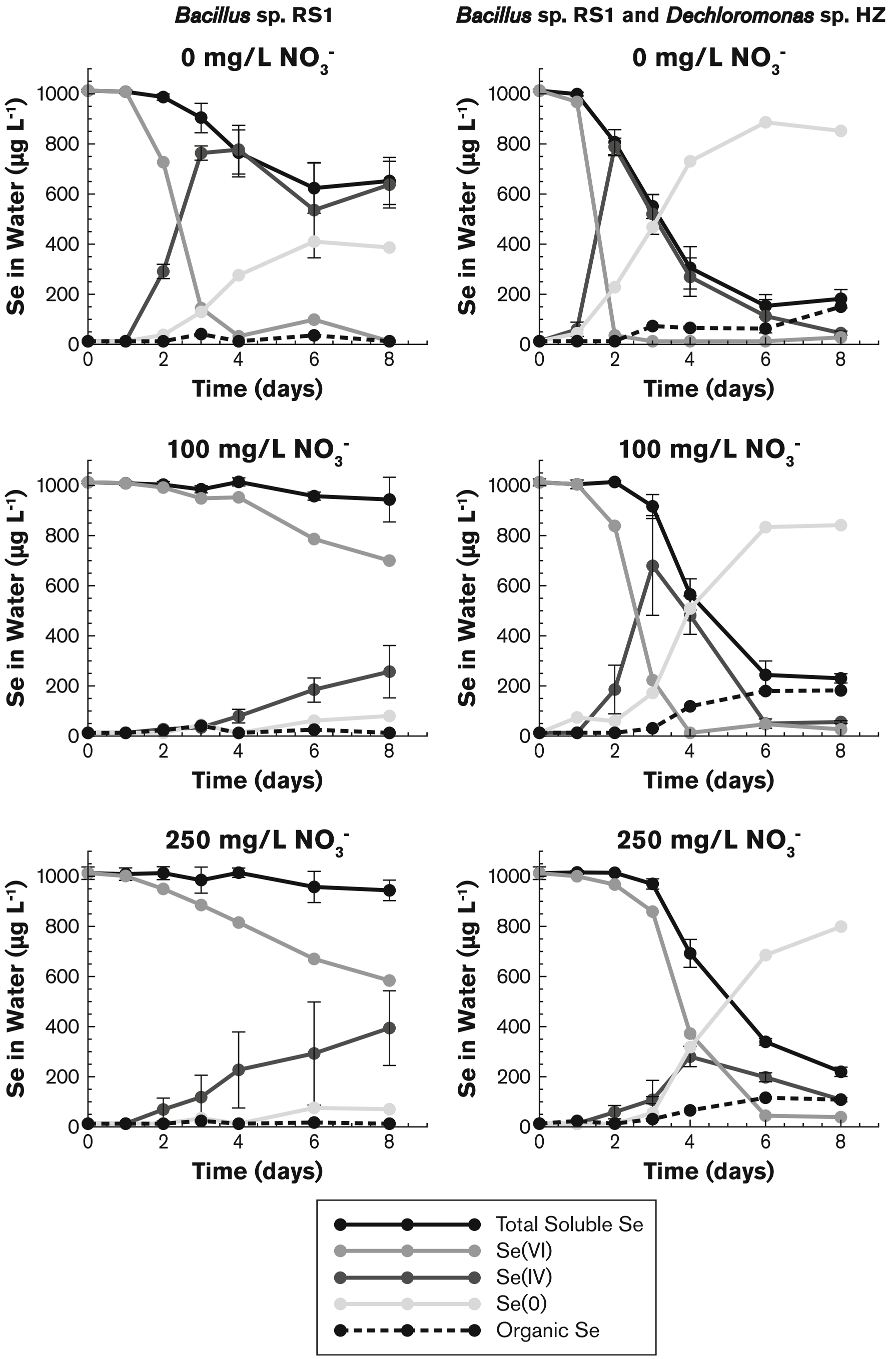

Fig. 7.5 Se(VI) reduction by *Bacillus* sp. RS1 and *Dechloromonas* sp. HZ (*right panel*) and *Bacillus* sp. RS1 *alone* (*left panel*) in synthetic drainage water containing in the 0, 100, and 250 mg L^{-1} NO_3^-. Error denotes standard deviation of three observations

When the NO_3^- concentrations were 100 and 250 mg NO_3^- L^{-1}, the rate of reduction of Se(VI) in the drainage water by *Bacillus* sp. RS1 slowed down. In 8 days, Se(VI), Se(IV), and Se(0) accounted for 56.5–68.6, 24.7–37.7, and 5.92–7.18 % of the added Se, respectively, when the NO_3^- concentration in the AD water varied from 100 to 250 mg NO_3^- L^{-1}. Additions of *Dechloromonas* sp. HZ significantly increased removal of Se(VI) compared to those inoculated only with *Bacillus* sp. RS1 in drainage water containing 0, 100, and 250 mg NO_3^- L^{-1}. In 8 days, 98–99 % of the added Se(VI) was reduced to Se(0), with small amounts in Se(IV) and Se(-II) formed (Zhang and Frankenberger 2007). The presence of *Dechloromonas* sp. HZ also enhanced the removal of NO_3^- in the treated water. Therefore, employing *Dechloromonas* sp. HZ to reduce NO_3^- in AD water would enhance the effectiveness Se(VI)-reducing bacteria in removing Se(VI), although *Dechloromonas* sp. HZ by itself did not reduce Se(VI).

Bacillus sp. RS1 not only reduced Se(VI) in the synthetic AD water, but also effectively reduced the Se(VI) present in the natural AD water that was treated with 0.1 % molasses (Fig. 7.6). The natural AD water was collected from the western SJV, California. The water contained 782 μg L^{-1} Se(VI) and 191 mg L^{-1} NO_3^- with pH of 8.2 and EC of 11.56 dS m^{-1}. In aseptic incubation, the Se(VI) concentration of the AD water changed little, from 784 to 748 μg L^{-1}. In the presence of *Bacillus* sp. RS1, 81 % of the added Se(VI) in the AD water was reduced. When the two bacteria species, *Dechloromonas* sp. HZ and *Bacillus* sp. RS1, were used in combination, the Se(VI) in the natural AD water was almost entirely reduced by in 6 days. Combining *Bacillus* sp. RS1, a Se(VI) reducing bacterium, with *Dechloromonas* sp. HZ, a NO_3^- reducing bacterium, and using molasses as the electron acceptor, Se(VI) in AD water containing relatively higher levels of salts and NO_3^-, may be effectively reduced (Zhang and Frankenberger 2007).

7.3.2 Zero-Valent Iron for Bacterial Reduction of Se(VI)

Zero-valent iron (ZVI) is a moderately strong reducing agent that has been used to remove Se(VI) from water through reductive and adsorptive processes (Murphy 1988; Zhang et al. 2005; Zingaro et al. 1997). For example, green rust, one of the Fe oxyhydroxides, can serve as a reducing agent to abiotically reduce Se(VI) to Se(IV) and Se(0) (Myneni, et al. 1997; Refait et al. 2000). Ferrihydrite and goethite are strong adsorbents that can be used to effectively remove Se(IV) from water (Balistrieri and Chao 1987, 1990). Addition of ZVI enhanced the bioreduction Se (VI) to Se(0) in water (Zhang and Frankenberger 2006).

We demonstrated the removal of Se(VI) from a strongly saline drainage water collected from the western SJV, California (Fig. 7.7). The saline drainage water contained 338 μg L^{-1} Se(VI) and 34.1 mg L^{-1} NO_3^--N with an EC of 17.3 dS m^{-1} and pH of 7.4. The Se(VI) concentration changed little in the drainage water in aseptic conditions and was slightly reduced from 338 to 281 μg L^{-1} if the drainage

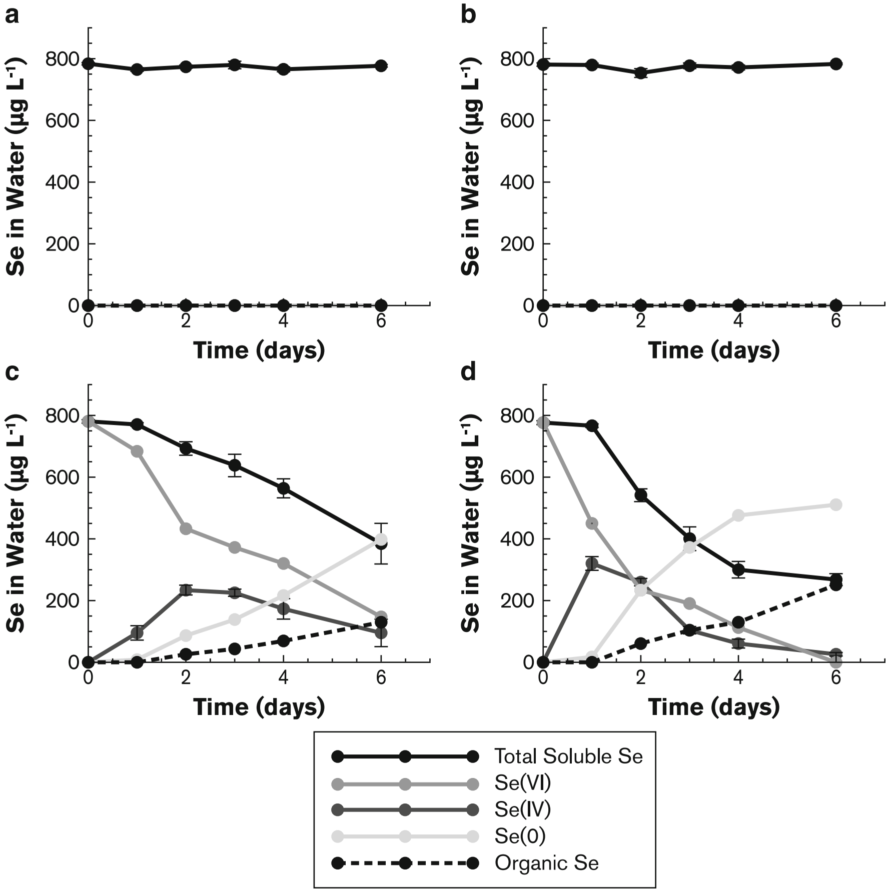

Fig. 7.6 Bio-reduction of Se(VI) in natural agricultural drainage water: (**a**) bacteria- and substrate-free control, (**b**) substrate free control with *Bacillus* sp. RS1, (**c**) 0.1 % molasses substrate with *Bacillus* sp. RS1, and (**d**) 0.1 % molasses substrate with *Bacillus* sp. RS1 and *Dechloromonas* sp. Error denotes standard deviation of three observations

water was inoculated with *C. braakii*. Adding 5 g L^{-1} of 40–60 mesh ZVI into the drainage water that was inoculated with *C. braakii* significantly enhanced the removal of Se(VI) from 338 to 32.9 µg L^{-1} in 7 days. In contrast, the Se (VI) decreased only slightly, from 338 to 293 µg L^{-1} in the drainage water when treated solely with ZVI.

The O_2 produced in the AD water under the Se-reductive process may be rapidly removed by the ZVI via the following reaction: $2Fe^o + 2H_2O + O_2 \rightarrow 2Fe^{2+} + 4OH^-$, keeping an anaerobic environment and enhancing the reduction of Se(VI) to Se(IV), and then to Se(0) by *C. braakii*. Adsorption of Se(IV) to Fe oxyhydroxides formed from the oxidation of ZVI also enhances the removal of Se.

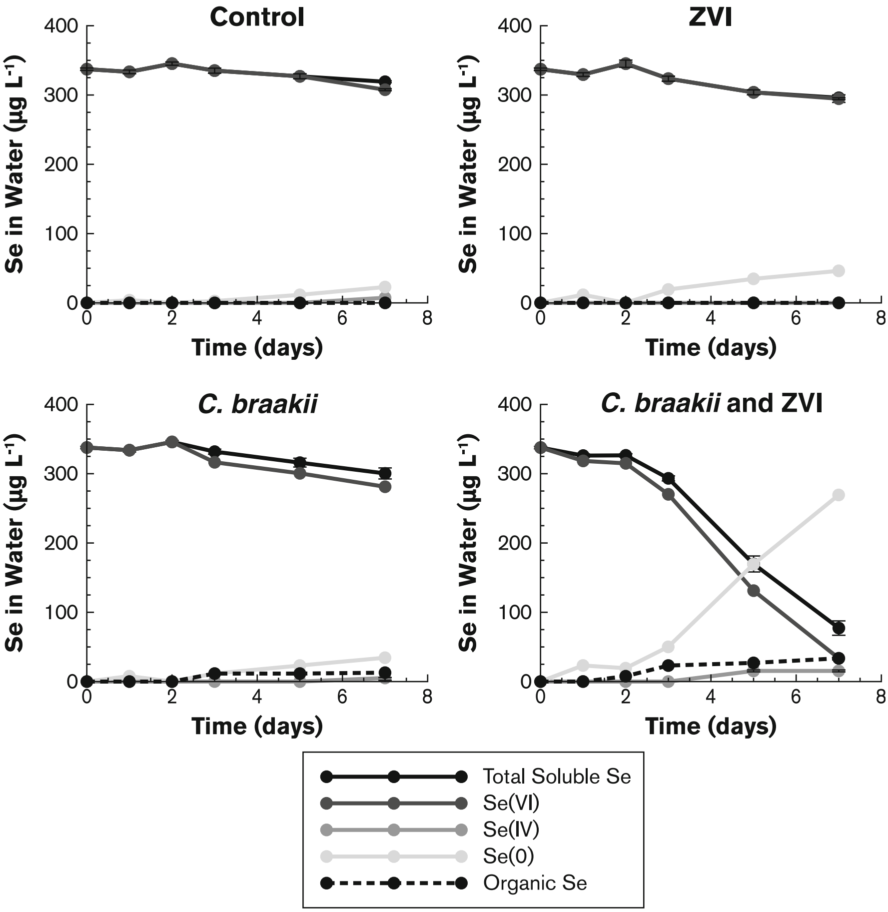

Fig. 7.7 Se removal in 0.1 % molasses amended saline agricultural drainage water of San Joaquin Valley under varying experimental conditions: Control denotes aseptic control and amendment free culture, *C. braakii* denotes culture inoculated with 1 mL of washed *C. braakii* cell suspension, ZVI denotes culture treated with 1 g of 40–60 mesh zero valence iron, and *C. braakii* and ZVI denotes culture inoculated with 1 mL of washed *C. braakii* cell suspension and treated with 1 g of 40–60 mesh zero valence iron. Error denotes standard deviation of three observations

7.3.3 Redox Mediator for Bacterial Reduction of Se(VI) to Se(0)

Humic substances (HS) are ubiquitous in terrestrial and aquatic environments. The redox-active quinone moieties have been shown to play important roles as electron carriers to stimulate the reductive biotransformation of azo dyes, nitroaromatic contaminants, and polyhalogenated contaminants by shunting electrons between an external electron donor and the contaminants (Field 2001). For example, anthraquinone-2,6-disulfonate (AQDS) can increase the reaction rate by acting as a redox mediator that shifts electrons between its oxidized, quinone from AQDS and its reduced, hydroquinone from AHQDS (anthrahydroquinone-2,6-disulfonate) (Field et al. 2000). van der Zee et al. (2001) reported that the first-order rate constant for the reduction of reactive red 2 (RR2) dye by methanogenic granular sludge could be increased seven-fold when AQDS was added into the RR2 containing wastewater. Coates et al. (2002) revealed that bacteria can use AHQDS directly, as an electron donor, to reduce NO_3^- to N_2 with acetate as a carbon source. The redox potential of NO_3^-/N_2 in an aquatic system is very similar to that of Se (VI)/Se(IV). Therefore, AHQDS may be used by Se(VI)-reducing bacteria as an electron mediator to accelerate reduction of Se(VI) to Se(0) in aquatic systems. The culture medium used in our study (Losi and Frankenberger 1997) contained K_2HPO_4, 0.225 g; KH_2PO_4, 0.225 g; $(NH_4)_2SO_4$, 0.225 g; NaCl, 0.46 g; $CaCl_2.2H_2O$, 0.005 g; $MgSO_4$, 0.117 g; $FeCl_2$, 0.0005 g; glucose 0.5 g; yeast extract, 0.25 g, and 1 mL of trace element solution in each liter of solution. After adding AQDS, ranging from 0 to 250 mg L^{-1}, the medium was autoclaved and 2,000 μg L^{-1} of Se(VI) was added prior to use (Zhang et al. 2007b).

In the un-inoculated control, there was little change in the Se(VI) concentration in 8 days (Fig. 7.8). In the presence of *E. taylorae*, the Se(VI) in the culture medium, amended with different amounts of AQDS, was reduced almost entirely. In the medium without the addition of AQDS, the Se(IV) concentration decreased from 2,000 to 890 μg L^{-1}. In contrast, the addition of 50 and 250 mg L^{-1} AQDS significantly enhanced Se(IV) reduction from 2,000 to 250 and 47 μg L^{-1}, respectively, in 8 days. The resultant Se(0) amounted to 1,080, 1,700 and 1,820 μg L^{-1}, respectively, in the media containing 0, 50, and 250 mg L^{-1} of AQDS.

The Se(IV) was reduced more effectively in the medium amended with 250 mg L^{-1} AQDS than without addition of AQDS, resulting in 97 vs. 52 % reduction of Se(IV) to Se(0) under comparable conditions. The *E. taylorae* used an organic carbon source and reduced AQDS to AHQDS. When the Se(VI) was being reduced, the AHQDS became the electron donor, allowing the reaction to proceed and thus enhancing the Se(IV) reduction. The AQDS however had little effect on reduction of Se(VI) in the culture medium, while Se(IV) was reduced rapidly to Se(0).

Although *E. taylorae* is unable to employ AHQDS as a redox mediator to enhance Se(VI) reduction, it is capable of using AHQDS as an electron donor in the reduction of Se(IV) to insoluble Se(0). Rau et al. (2002) reported that different hydroquinone/quinone couples had redox potentials ranging from – 350 to 280 mV. If a quinine of appropriate redox potential range is selected, it is possible that the bacteriological reduction of both Se(VI) and Se(IV) could be accomplished and would make bioremediation of Se-contaminated AD water more practical.

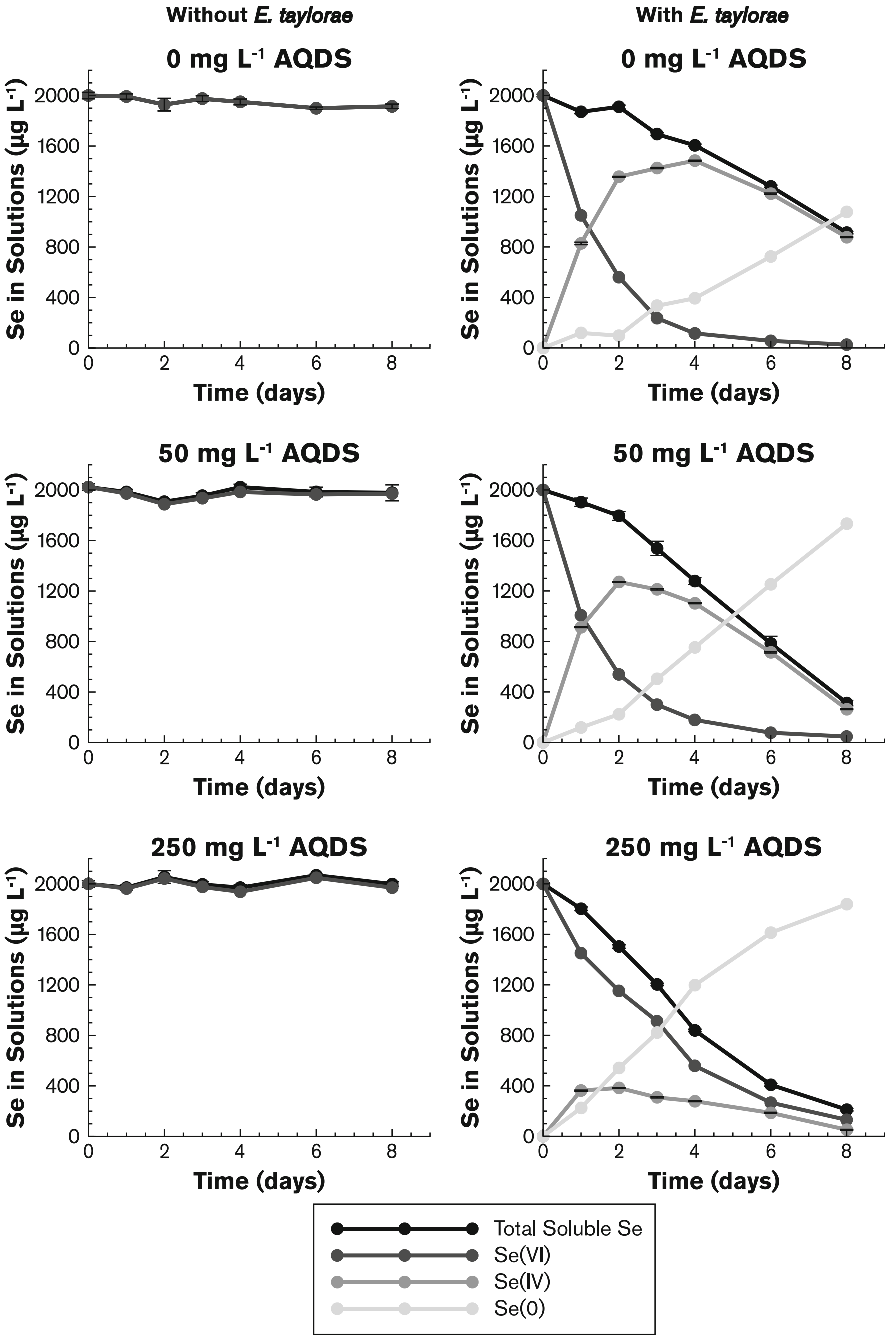

Fig. 7.8 Redox mediator, anthraquinone-2,6-disulfonate (AQDS), on enhancing Se(VI) reduction to Se(0) in synthetic drainage water containing 2,000 µg L^{-1} Se(VI) and 250 mg L^{-1} yeast extract and with and without inoculation of *E. taylorae*. Error denotes standard deviation of three observations

7.4 Removal of Se(VI) in Agricultural Drainage Water – Pilot Tests

7.4.1 Rice Straw

Rice straw is abundant in the SJV, as the Sacramento – San Joaquin Delta is a rice production region, and it may be used as a source of carbon and nutrients for bacteria engaged in Se reduction. Rice straw contains 30–35 % carbon and significant quantities of K, P, N, S, Na and Mg that, upon soaking, may be released into the solution, becoming substrate for culturing microorganisms (Jenkins et al. 1996). Mikkelsen et al. (1989) showed that when Se(VI) was added to flooded, rice-growing fields, it could be reduced completely in 10 weeks. Both *E. taylorae* and *Bacillus* sp. RSl, Se-reducing bacterial strains first isolated from rice straw (Table 7.1), are effective in reducing Se(VI) to Se(0). The rice straw therefore would be a logical reservoir of Se(VI)-reducing bacteria and a substrate of carbon and nutrients in sustaining the microorganisms. In the SJV, rice straw could be an inexpensive aid to *in situ* bioremediation of Se-contaminated AD water, if its effectiveness can be demonstrated on a larger scale (Zhang and Frankenberger 2003c).

The bench top pilot flow-through bioreactor channel system (BCS) designed to remove Se(VI) from AD water through reduction to Se(0) and to trap Se(0) with rice straw was assembled (Fig. 7.9) by linking three simulated flow through channel sections (made of 500-ml semi-transparent plastic bottles laying horizontally) in serial position and placing 10, 10, and 50 g of rice straw inside, respectively. At the channel outlet, a 60-mL plastic bottle served as the sampling site (SS) of effluent before flowing into the next section. When soluble Se(VI) was pumped out of the reservoir and flowed through the channel sections, Se(VI)-reducing bacteria used Se(VI) in the water as a terminal electron acceptor for anaerobic respiration and used the organic matter in rice straw as an electron donor (Zhang and Frankenberger 2003a, b). Sections 7.1 and 7.2 of the simulated channel sites was where Se(VI) reduction and Se(0) precipitation took place. Section 7.3 was designed as a strong reduction and filtration zone where soluble Se(VI) in the effluent of the previous section could be reduced further to Se(0) and then the Se (0) in suspension could be trapped.

At the beginning, the simulated AD water free of NO_3^- was flowing through the BCS at a rate of 250 mL day^{-1}. Each liter of the simulated AD water had the following constituents: 5.645 g Na_2SO_4, 1.486 g $MgSO_4$, 0.867 g NaCl, 0.413 g $NaHCO_3$, 2.017 g $CaCl_2 \cdot 2H_2O$, and 0.00244 g Na_2SeO_4. The final EC and pH of the solution were 10.4 dS m^{-1} and 8.1, respectively. The total soluble Se concentration in the effluent of the first flow-through section obtained at sampling site SS1 ranged from 1,007 to 1,030 μg Se L^{-1} during the first 7 days. It decreased by one order of magnitude to 107 μg Se L^{-1} at day 11. As the reactor was operated at the steady state condition from days 13 to 92, total soluble Se concentrations of the effluents fluctuated in a range of 50.7–206 μg Se L^{-1}.

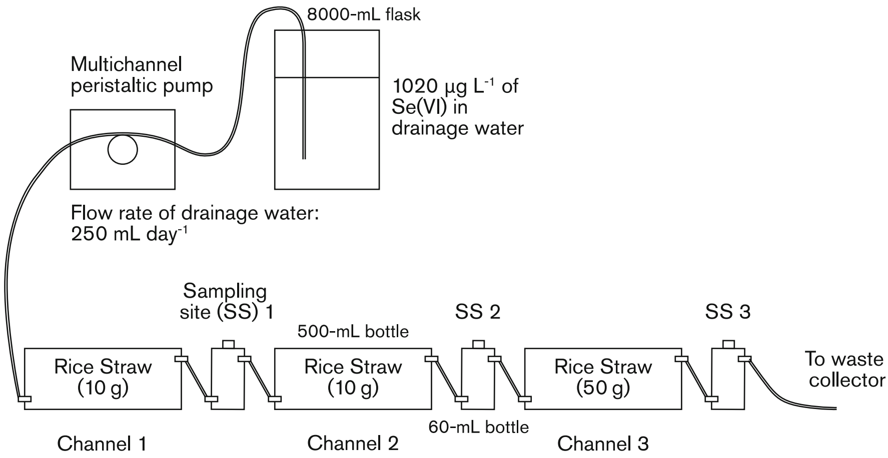

Fig. 7.9 Prototype flow through Se removal bioreactor

When drainage water containing 100 mg NO_3^- L^{-1} was deployed at day 92, the total soluble Se concentration increased to a range of 163–493 μg Se L^{-1} and suspended Se(0) ranged from 0 to 374 μg L^{-1} with an average of 112 μg L^{-1} in the effluent. At SS 2, total soluble Se in the drainage water was high in the first 8 days, ranging from 905 to 1,031 μg L^{-1}. It was less than 100 μg L^{-1} most of the time from days 13 to 92. Total soluble Se increased to a range of 70–226 μg L^{-1} from day 92 to the end of the experiment. Un-precipitated Se(0) ranged from 0 to 335 μg L^{-1}, with an average of 70 μg L^{-1}. At SS 3, total soluble Se in the drainage water decreased from 1,010 to 72.2 μg L^{-1} in the first several days. It ranged from 15.4 to 62.9 μg L^{-1}, with an average of 29.4 μg L^{-1} during the rest of the experiment. Un-precipitated Se(0) ranged from 0.11 to 118 μg L^{-1}, with an average of 33.9 μg L^{-1}.

During 122 days of pilot testing, a large amount of Se(VI) was removed from the drainage water in the flow-through BCS. The primary form of reduced Se in the BCS reactor was Se(0), as evident by the red precipitates at the bottom, on the inner wall of the simulated flow-through channels, and on the surface of rice straw. The mass balance showed that in 122 days, about 90 % of the Se(VI) input was reduced to Se(0) precipitate (Table 7.2). Of the Se(0) precipitate formed, 68.5, 11.6, and 6.91 % were deposited in channel Sects. 7.1, 7.2, and 7.3, respectively, and 3.13 % of the Se(0) was found in the final effluent.

The loss of rice straw due to biodegradation was 3.9, 3.1, and 9.9 g for 10, 10, and 50 g of rice straw deposited in channel Sects. 7.1, 7.2, and 7.3, respectively. In this pilot setup, the only substrate added was rice straw. Each unit weight loss of rice straw resulted in 0.0535, 0.0114, and 0.021 s unit of Se captured in channel Sects. 7.1, 7.2, and 7.3, respectively. If properly scaled up, the flow-through Se bioreduction channels would be effective to remove Se, as the AD water works its way through the conveying system to a terminus collection point, such as an evaporation pond, reuse location, and desalting installation.

Table 7.2 Mass balance of Se and rice straw in bench top flow through Se bio-reduction pilot plant following 122 days of testing

Channel section	Se input (mg)	Se (0) trapped (mg)	Soluble Se retained (mg)	Soluble Se effluent (mg)	Se(0) in effluent (mg)	Rice straw decomposed (g)	Se removal (mg Se/g straw)
1	30.5	20.855	0.361	–	–	3.9	5.35
2	–	3.53	0.157	–	–	3.1	1.14
3	–	2.109	0.039	1.569	0.955	9.9	0.21

7.5 Summary

Selenium reductions from Se(VI) to Se(0) are naturally occurring biological processes; however, they are often limited by the lack of appropriate microbial populations, the redox environment, substrates and nutrients, and neutralizing competitive ions, such as nitrate. The AD water generated in the SJV encountered all of the above-mentioned obstacles to natural Se reduction. In addition, the volume of the Se-borne AD water is enormous and widespread.

We have demonstrated that Se in the AD water generated in the SJV may be reduced, precipitated, and thus removed from the water stream. We also tested and illustrated how the bio-reduction of Se may be enhanced and thus optimized by manipulating external factors, such as bacterial strains, electron acceptors, and redox reaction enhancers. In a large-scale, regional approach to treatment in the field, cost and efficiency would always be considerations. Our laboratory-scale research and pilot study have accomplished the following objectives:

- Isolated and identified a slate of Se(VI)-reducing bacteria from different environments that may be deployed in the bioremediation of AD water.
- Demonstrated that readily available, inexpensive organic substrates, such as molasses and rice straw, are effective electron acceptors for reducing soluble Se(VI) in AD water to insoluble Se(0) within days, not months or years, when the AD water is inoculated with the identified and tested Se(VI)-reducing bacteria, such as *Enterobacter taylorae*.
- Tested and demonstrated that NO_3^- ion, prevalent in AD water and a competitor of Se(VI) as an electron acceptor in the same aquatic system, may be neutralized by introducing both *Bacillus* sp. RS1, a Se(VI)-reducing bacterium, and *Dechloromonas* sp. HZ, a nitrate-reducing bacterium, into the aquatic system.
- Showed that the appropriate redox environment favorable to Se reduction might be maintained with abiotic chemical redox mediators, such as zero valent iron (ZVI) and redox-active quinone moieties of anthraquinone-2,6-disulfonate, in the presence of AD water inoculated with the identified Se(VI)-reducing bacteria, such as *Citerobacter freundii*.

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Chapter 8
Chemical and Biological Processes of Evaporation Ponds

Suduan Gao and Andrew C. Chang

Agricultural evaporation ponds began to be constructed in the San Joaquin Valley in the early 1970s at about the same time as construction of Kesterson Reservoir, a wildlife refuge and terminal holding pond to store saline subsurface drainage waters conveyed by the San Luis Drain. Since the eco-toxicological crisis occurred at the Kesterson Reservoir in 1983, the chemistry and biology of evaporation ponds have been the focus of extensive investigations, especially the transformations of trace elements, such as selenium (Se). Agricultural evaporation ponds are employed to impound and dissipate saline agricultural drainage water in irrigated areas lacking opportunities for offsite disposal, such as the San Joaquin Valley. They provide an option for in-basin management of the salts and Se generated by irrigated crop production. Drainage waters in these ponds are desiccated by solar evaporation, resulting in elevated levels of dissolved mineral salts and trace elements, along with precipitation of evaporite minerals, a process referred to as evapoconcentration, hereafter. As the salts and trace elements accumulate in the ponds and undergo biogeochemical processes in the aquatic ecosystem, they may pose a hazard to wildlife attracted to the ponds for foraging and nesting. Seepage losses may also degrade the quality of adjacent surface water and groundwater bodies.

Evaporation ponds are a stopgap, interim management response to extend the period of productive agriculture, and their temporal scale depends on a variety of factors, primarily the success of on-farm efforts to reduce the volume of saline drainage (see Chaps. 10 and 11). The evaporation ponds and other in-basin salt

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DOI 10.1007/978-94-007-6851-2_8, © Springer Science+Business Media Dordrecht 2014

management approaches have a finite utility and projected life because they do not provide for net export of salts brought about by importing water to satisfy irrigation demands.

8.1 Status of Evaporation Ponds in the San Joaquin Valley

Most of the evaporation ponds in the San Joaquin Valley were constructed between 1972 and 1985 and, at their peak, 3,000 ha of evaporative surface area was operational (Ford 1988; Tanji et al. 1993). Many of the ponds were installed in the southern half of the Valley, the hydrologically closed Tulare Basin. Each facility is comprised of multi-cells flow through systems with surface areas ranging from a few up to 750 ha per cell. Water typically flows in a serial order from one cell to another, eventually reaching a terminus cell. Evaporation and seepage are the main losses of water. Evapotranspiration (ET) rates of this region range from 1,400 to 1,600 mm year^{-1} (CIMIS 1999) with the highest potential ET (ET_0) up to 240 mm month^{-1} in the summer and the lowest ET_0 of 24 mm month^{-1} in the winter. Pond evaporation rates are roughly 1.5 m year^{-1}, significantly exceeding the area rainfall of about 200 mm year^{-1} that occurs primarily between October and April (Grismer et al. 1993). The majority of evaporation ponds were constructed on clay soils. Seepage rates decreased dramatically within the first 1–2 years of pond operation due to "bottom sealing" from microbial activity in pond bed materials and were on the order of a few mm day^{-1} after the pond stabilized.

Each pond system received subsurface tile drainage from a designated area and the inflow was not regulated. For example, the evaporation pond system of a 22,700 ha of tile-drained fields received an annual volume of 39 million m^3 (Tanji et al. 1993). The State Water Resources Control Board (SWRCB) requires monitoring of evaporation ponds in terms of water quality and sediments under Title 22 of the California Code of Regulations. The soluble threshold limit concentration (STLC) and the total threshold limit concentration (TTLC) of sediments are used to determine whether the contents in the ponds should be categorized as hazardous wastes. The STLC for Se, As (arsenic), B (boron), and Mo (molybdenum) are 13, 67, 6,476 and 3,648 μM, respectively, and the TTLC for Se and As are 10,000 and 50,000 mg kg^{-1}, respectively, as stipulated in Title 22 of California Code of Regulations. If pond waters or sediments of an evaporation basin exceed the thresholds, the basin is subject to temporary or permanent closure, according to the hazardous waste management criteria.

Due to the high costs of meeting regulatory requirements, some of the ponds were closed, and some have been converted to a component of the integrated on-farm drainage management (IFDM) systems, described in Chap. 11. By the end of 2007, eight evaporation pond systems remained, representing 28 % of those in use in 1988. Four of the eight were pond systems located in the Tulare Lake bed, two were located in the alluvial fan region, and one located in the basin trough, covering a total surface area of 1,930 ha, 65 % of the total capacity in 1988 under

Table 8.1 Geometric mean trace element concentrations of water in evaporation ponds, San Joaquin Valley (Chilcott et al. 1993; Chilcott and McVay 1993; Westcot et al. 1993)

	Water (g L^{-1})				Sediment (mg kg^{-1})		
Element	San Joaquin Valley	Alluvial fan	Basin trough	Tulare Lake Bed	Alluvial fan	Basin trough	Tulare Lake Bed
B	25,000	60,000	27,000	21,000	140	110	86
As	101	19	22	210	6.0	4.9	13
Se	16	380	3	9	3.8	0.2	0.4
Mo	2,817	1,500	640	1,500	0.7	2.5	6.1
U	308	370	120	360	4	7	12
V	22	20	19	22	63	59	58

five operators. These ponds will continue to play an important role in drainage disposal to sustain agricultural productivity until a permanent solution can be found.

8.1.1 Water Quality and Sediments

The evapoconcentration process leads to formation of salts or mineral precipitates (commonly halite, thenardite, and mirabilite) and elevated concentrations of trace elements, primarily Se, B, As, Mo, U (uranium), and V (vanadium) in pond water (Tanji 1990a; Chilcott et al. 1993; Chilcott and McVay 1993; Westcot et al. 1993; Ong et al. 1997). The solution concentrations of trace elements in the evaporation ponds were much higher than their background concentrations in seawater and in natural saline-sink lakes in the western USA. The accumulation of trace elements in evaporation ponds was alarming, considering the average age of the ponds was <15 years (Westcot et al. 1993). Relatively speaking, higher B and Se levels were found in the alluvial fan region and higher As levels were found in the Tulare Lake bed (Table 8.1), representing the levels of these trace elements present in agricultural fields in different physiographic areas (Westcot et al. 1993). The soluble Mo, U, and V levels of the ponds appear similar in the two regions.

There have not been comprehensive accounts of chemical and minerals in evaporation ponds throughout the San Joaquin Valley since 1993 (Table 8.1), except for limited research data on selected evaporation ponds. In an evaporation basin facility in the Tulare Lake Drainage District (TLDD) in Tulare Basin, monthly sampling and analysis from 1985 to 1995 indicated that inlet water had median As and Se concentrations of 97 and 2 μg L^{-1}, respectively, and pond waters ranged from 110 to 420 μg As L^{-1} and 21–29 μg Se L^{-1}, respectively (Fujii 1988). The 2004–2005 measurements from the same facility showed that As concentrations increased from 120 at the inlet up to 1,000 μg L^{-1} in the terminus pond, while Se concentrations of the terminus pond water ranged from 7 to 10 μg L^{-1}, which were lower than that at the inlet, 15 μg L^{-1} (Gao et al. 2007a). The fluctuating solution concentrations of Se and As in evaporation ponds indicate that the food chain in the aquatic ecosystem could play a significant role in the biogeochemical processes.

Data reported in the early 1990s indicated that concentrations of B, Se, and As in some ponds approached or exceeded the STLC (Ong et al. 1995). Although sink-mechanisms were identified for some of the trace elements, there had not been any reported cases that extractable trace elements in the sediments exceeded the TTLC (Ong et al. 1995, 1997). Selenium has been documented to accumulate significantly in sediments (Zawislanski and Zavarin 1996; Tokunaga et al. 1996). Although As has some sink mechanisms (Ong and Tanji 1993), a recent finding indicated its conservative properties in pond waters (Gao et al. 2007b). Molybdenum, U, and V were also considered to have some sink mechanisms, while B was considered to be a conservative element (Westcot et al. 1993; Ong et al. 1995). Earlier studies indicated that trace element accumulation tended to occur in the organic rich layer of the soil, within 6 cm of the pond's bottom (Chilcott et al. 1993). Trace element (As, Mo, and V) concentrations were highest in the top 2–4 cm of sediment in a vegetated wetland and decreased with depth (Fox and Doner 2003). There was no clear trend in the Se and As concentrations in evaporation pond sediments in recent investigations when 25-cm sediment cores were analyzed at 5-cm increments (Gao et al. 2007a, b).

8.1.2 Salinity and General Water Chemistry

In natural water, salinity is caused primarily by the presence of Na^+, Ca^{2+}, Mg^{2+}, K^+, Cl^-, HCO_3^-, and SO_4^{2-} ions. For measurement purposes, salinity may be expressed as a collection of dissolved salts or ionic concentrations for each individual ion, ion pairs, or ion complexes, stated in terms of electrical conductivity (EC) induced by soluble ions present in water (EC, dS m^{-1} or dS cm^{-1}), or, collectively, as the sum of mass concentrations of total dissolved salts (TDS, mg L^{-1}). The evapoconcentration process results in a dramatic increase in the salinity of water in the evaporation ponds over time. For an evaporation pond system receiving inflow drainage of 10 dS m^{-1}, the water in the ponds varied from 22 to 90 dS m^{-1} in 1985–1986 (Fujii 1988) and had risen up to 120 dS m^{-1} in 2004 (Gao et al. 2007a). In extreme cases, an EC increase up to 178 dS m^{-1} had been recorded (Ong et al. 1995). The geometrical mean of TDS of pond waters was 31,000 mg L^{-1} (Westcot et al. 1993). As evaporation continues, the concentrations can reach the solubility product constant (K_{sp}) of selected mineral salts and precipitate out. The evapoconcentration factor (ECF) was employed, based on the conservative properties of Cl ions in water (Tanji 1990a). The ECF denotes the ratio of $[Cl^-]_{pond\ water}/[Cl^-]_{inlet\ water}$ which is a valid indicator of the evapoconcentration process until the Na^+ and Cl^- concentrations of the water reach the K_{sp} of halite and precipitation occurs. In multi-cell drainage evaporation ponds, the ECF increases along the water flow path (Tanji 1990a; Gao et al. 2007a). The highest ECF observed was about 20, with a corresponding EC of 120 dS m^{-1} (Gao et al. 2007a).

Because of stoichiometry, the chemistry of salts in evaporation basins may be examined by chemical speciation modeling or by correlating concentrations of individual soluble ions to EC (or ECF) and to the composition of salt minerals. In evaporation ponds, the soluble constituents Na^{+}, Cl^{-}, and SO_4^{2-} are the highest in mass concentration (Fig. 8.1) (Westcot et al. 1993; Ong et al. 1995; Gao et al. 2007a). The mass concentrations of soluble Mg^{2+} and Ca^{2+} are an order of magnitude lower, followed by HCO_3^{-} and K^{+} concentrations. Magnesium (Mg^{2+}) concentrations are consistently higher than Ca^{2+} concentrations and more linearly correlated to EC or Cl^{-} concentrations. Figure 8.1 illustrates some of the correlations among soluble constituents and the salinity increase indicated by the Cl^{-} concentration. The pH of the pond waters ranged from about 8 to above 9, reflecting a carbonate-dominated aqueous system.

As the water evapoconcentrated in the ponds, the concentration of the dissolved minerals initially would increase in proportion with the water's EC as it gained in ionic strength (Fig. 8.1). When the EC of pond water exceeded 120 dS m^{-1} or the ECF of the evapoconcentration process exceeded 20, dissolved ions began to form complex ion pairs or precipitates and the plotted data deviated from the earlier linear trends. Figure 8.1 reflects the stoichiometry of the reactions. When the concentrations of dissolved ions in the water exceeded the solubility product constants (K_{sp}) of the minerals, precipitation occurred. Calcite, being the lowest in K_{sp}, would most likely be the first mineral to precipitate (Tanji 1990b). However, the amounts of precipitated minerals formed were also dependent on the ionic compositions of the water. Analyses of mineral solids collected from evaporation ponds indicated that the evaporites were dominated by the following constituents: Na^{+} and SO_4^{2-}, variable levels of Cl^{-}, and low levels of Ca^{2+} and Mg^{2+} of which the pond waters were either predominately $Na^{+} - Cl^{-}$, $Na^{+} - SO_4^{2+}$ or $Na^{+} - Cl^{-} - SO_4^{2+}$ water (Ong et al. 1997). Smith et al. (1995) developed a brine chemical equilibrium model and predicted that the most common evaporites in the early evapoconcentration process are calcite ($CaCO_3$) and gypsum ($CaSO_4 \cdot 2H_2O$) followed by thenardite (Na_2SO_4) and halite (NaCl). Although calcite and gypsum have low K_{sp} and are likely to be formed, the low levels of Ca^{2+} limit the quantity of these minerals. The most common minerals actually observed in the evaporation ponds in the San Joaquin Valley were halite, thenardite, and mirabilite ($NaSO_4 \cdot 10H_2O$) (Tanji et al. 1992). Other minor minerals that were expected include gypsum, calcite, bloedite ($Na_2Mg(SO_4)_2 \cdot 4H_2O$), glauberite ($Na_2Ca(SO_4)_2$) and nesquechonite ($MgCO_3 \cdot 3H_2O$) (Westcot et al. 1993; Smith et al. 1995). Smith et al. (1995) predicted that calcite and gypsum may form precipitates at an ECF < 20; glauberite and thenardite began to form at an ECF around 20; and at an ECF greater than 20, thenardite was the dominant precipitate along with glauberite and bloedite.

Among the trace elements, the concentrations of B and As also exhibited a linear proportional increase with the Cl^{-} concentration and EC of the pond water, while the Se concentration did not exhibit any increase and sometimes was lower than the Se concentration of the inflow, due to its biochemical transformations. Boron concentrations were orders of magnitude higher (in the mg L^{-1} range) in

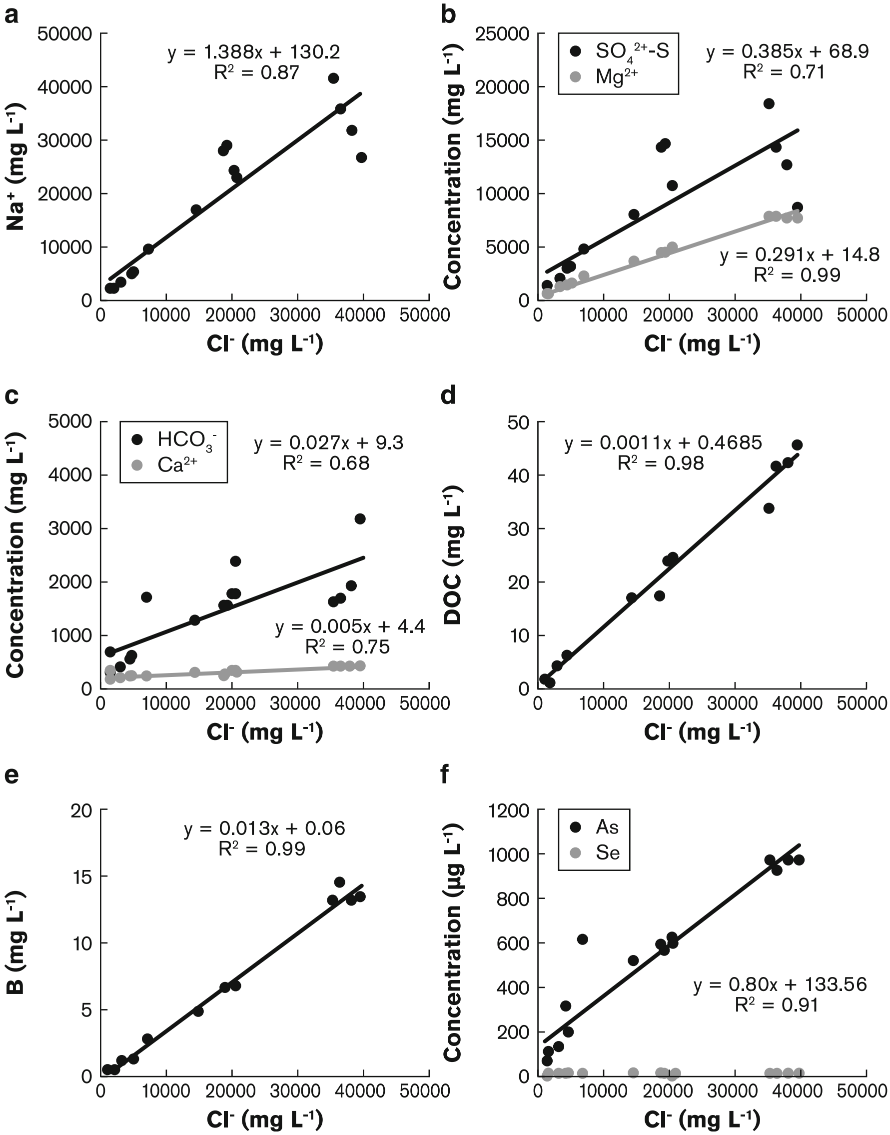


Fig. 8.1 Correlation of selected chemical constituents of water in evaporation ponds: (**a**) Na^+, (**b**) SO_4^{2-}-S and Mg^{2+}, (**c**) Ca^{2+} and alkalinity, (**d**) DOC, (**e**) B, and (**f**) As and Se with respect to that of Cl^- which is assumed to be a conservative parameter influent drainage water (Modified from data in Gao et al. 2007a)

comparison to those of other trace elements (all in the μg L^{-1} range). The partitioning of trace elements into evaporite minerals does not take place in the evapoconcentration process because of their relatively low concentrations, ~100 times depleted, relative to the solution phase (Ong et al. 1997).

Although nitrogen (N) and phosphorus (P) were much lower in concentration (<1 to a few mg L^{-1}) than the dissolved salts in the evaporation ponds, they play an important role in regulating aquatic biological activities. The N content of inflowing drainage water, mostly in the form of nitrate (NO_3^-), can be considerable. In an evaporation pond facility in the TLDD, the median concentration of NO_3^- was 20 mg N L^{-1} in 1985–1986 (Fujii 1988) and was maintained at about 6 mg N L^{-1} in 2004–2005 (Ryu et al. 2010). The decreased NO_3^-–N in the ponds indicated active biological and/or chemical NO_3^- reduction processes along the flow path, as evidenced by a relatively high ammonium (NH_4^+) vs. nitrate concentration in evaporation ponds (Ryu et al. 2010). The total P concentration in the same water decreased from about 1 mg P L^{-1} in the inlet to $\leq$0.2 mg P L^{-1} (Fujii 1988; Ryu et al. 2010). The biological roles of the evaporation ponds will be elaborated in later sections of this chapter.

8.1.3 Redox Chemistry and Transformation of Trace Elements

Vascular plants were not a featured component of evaporation ponds because they would have attracted wildlife habitation. However, indigenous algal and microbial activities were intense and resulted in biochemical transformation of trace elements (Ohlendorf et al. 1993). The anoxic conditions developed in the evaporation ponds triggered many redox reactions and affected trace elements' speciation, solubility, and aquatic toxicity. Selenium and As in the evaporation ponds have been studied more frequently than other trace elements (e.g., Tanji et al. 2006; Gao et al. 2007a, b; Ryu et al. 2010). Only limited information is available for Mo, U, and V (e.g., Amrhein et al. 1993, 1998; Duff et al. 1997a, b, 1999). Boron species in evaporation ponds are not subject to redox reactions and appear to have no significant sink mechanisms in the evapoconcentration process.

8.1.3.1 Redox Status of Evaporation Ponds

The anoxic conditions in the evaporation ponds took place in parallel with evapoconcentration of salts. The more concentrated were the salts, the more anoxic was the chemical environment. Water chemistry was examined along the flow path in a TLDD basin facility (Tanji et al. 2006; Gao et al. 2007a; Ryu et al. 2010). While solute concentrations increased along the flow path due to evapoconcentration, electron acceptors, such as O_2, NO_3^-, Fe(III) (ferric iron), and SO_4^{2-}, were successively consumed, and the redox potential changed gradually from the oxidizing to the reducing condition. The dissolved oxygen (DO) concentration near the bottom of the water column was significantly lower than that near the surface and the DO concentration of the water in the terminus cells was lower than that in the initial cell that received the fresh drainage water. The nitrate and Fe(III) concentrations of the water, decreased gradually along the flow path and were accompanied by increases in concentrations of NH_4^+ and Fe(II) (ferrous iron). When the water was saline, sulfide (S^{2-}) often was detected in the terminus cells. Under reducing conditions, the hypersaline water in the evaporation basins exhibited unusually high dissolved organic carbon (DOC) levels that showed a positive correlation with the EC of the water. The organic matter content in the surface sediments was found to be similar among cells (32–36 g kg^{-1}, on average). This increase in DOC with EC might have resulted from the serial transport of soluble salts and be due partially to the addition of chicken manure for enhancing the production of brine shrimp that had been active in some of the ponds for a number of years.

There were spatial variations in the redox conditions within a cell, but they were less pronounced, compared to the variations with respect to water depth and between cells (Tanji et al. 2006; Ryu et al. 2010). The pond was shallow, generally around 1 m deep. In the initial cells that received the inflow drainage water, the DO levels were lower toward the bottom of the water column immediately above the sediment layer, averaging 3.7 mg L^{-1}, as compared to the DO levels of water near the surface of the water column, which averaged 14.2 mg L^{-1}. For the surface layer, the DO levels of the initial cell were much higher than those in the terminus cell, 14.2 vs. 3.6 mg L^{-1}. Accompanying the decrease in DO levels were decreases in NO_3^- and Fe(III) concentrations and increases in Fe(II), dissolved manganese Mn (II), and S^{2-} concentrations in the pond's water. The S^{2-} concentration of the surface water ranged from the non-detectable level in the initial receiving cell to an average of 11 mg S^{2-} L^{-1} in the bottom layers of the terminus cell. As drainage water flows from the initial cell to the terminus cell, salts are concentrated, such that concentrations of oxidized species (NO_3^-, Fe(III), and DO) decline, and concentrations of reduced species (Fe(II), dissolved Mn (II), and S^{2-}) increase.

8.1.3.2 Selenium

Selenium is an element of special interest in the San Joaquin Valley due to its potential toxicity to aquatic life, especially to water birds and fishes (Ohlendorf et al. 1993). The national ambient water quality criterion for Se is 5 μg Se L^{-1} in surface water bodies (USEPA 1987). Based on the toxic effects of Se observed in aquatic biota, Hamilton and Lemly (1999) recommended that the threshold be lowered to 2 μg Se L^{-1}. The concentrations of Se in drainage water and evaporation ponds easily exceed the above-referenced aquatic toxicity thresholds, especially those located in the west side of the San Joaquin Valley where the soils are high in salinity and rich in seleniferous minerals, derived from parent materials affected by the sedimentary rocks from the Coast Ranges of California (Tanji et al. 1986). Weathering and sediment transport processes have resulted in widespread Se-rich soils and high Se concentrations in shallow groundwater, especially in the Panoche Fan areas (see Chap. 4). As the seleniferous minerals undergo further weathering and oxidation in the soil, the resulting soluble forms of Se are mobilized and are leached along with the drainage water when the land is irrigated.

In aqueous environments, the toxicity of Se is dependent on its chemical speciation. The most oxidized form, selenate (SeO_4^{2-} or Se(VI)), is readily soluble and is the dominant form of Se in agricultural drainage water in the San Joaquin Valley. Under mildly reducing conditions, Se(VI) will be reduced to selenite, (SeO_3^{2-} or Se(IV)), which has a higher affinity for adsorption to soil and mineral surfaces. Under highly reducing conditions, Se(VI) and/or Se(IV) will be reduced to elemental Se, (Se(0) or Se^0) and/or selenide, (Se(−II) or Se^{2-}). Elemental Se is a reddish precipitate and Se^{2-} includes metal selenide precipitates and organic Se (org-Se), some of which are volatile compounds. The above-described Se transformation processes occurred routinely in the agricultural drainage water evaporation ponds, as all of the chemical species have been found in water columns and/or sediment (Zhang and Moore 1996; Zhang et al. 1999; Gao et al. 2007b). The reductive processes are natural sinks for Se. Selenium in evaporation ponds may form solid phases and be separated from the water column or may form gaseous phases and be dissipated into the atmosphere. Under controlled environments, Se transformations may also be customized for mitigating Se-contaminated water (Quinn et al. 2000; Zhang and Frankenberger 2003; Higashi et al. 2003).

In evaporation ponds, drainage water flows sequentially through zones of incrementally more reduced conditions. Gao et al. (2007b) examined the Se concentrations and speciation of water in five evaporation ponds at two drainage water disposal facilities in the Tulare Basin. It should be noted that Se concentrations of drainage water in the Tulare Basin were much lower than those on the west side of the San Joaquin Valley. Nevertheless, the patterns established and the processes involved are the same. The drainage water entering the facilities had an average Se concentration of 16 μg Se L^{-1} and the water evaporated as it flowed through the ponds. Conservatively, the Se concentration of the pond water should be higher than that of the inflow. However, most pond waters had Se concentrations <10 μg Se L^{-1}, indicating that Se was being removed from the water columns when Se(VI) in the incoming water was sequentially reduced to Se(IV), Se(0), and org-Se. The chemical species of the inlet water was primarily Se(VI), with approximately 5 % as Se(IV), and a non-detectable level of org-Se. In selected ponds, as the soluble Se(VI) in water decreased, the reduced species of Se (IV) and org-Se increased to 40 and 27 % of the total Se in water, respectively. The formation of the more toxic, reduced Se species is a significant concern.

Microbes indigenous to the evaporation pond systems that were found and identified in pond sediments play important roles in reducing Se(VI) and Se (IV) species to Se(0) species (Siddique et al. 2005). Elemental Se was a significant end product of Se reductive processes in the evaporation ponds (Zawislanski and Zavarin 1996; Tokunaga et al. 1996) and accounted for 39–53 % of the total Se in the sediment (Gao et al. 2007b). Oremland et al. (1989) illustrated that bacteria were capable of dissimilatory Se(VI) reduction to Se(0) in anaerobic sediments. Dissimilatory microbial reduction of Se(0) in sediments to Se(−II) or metal selenides, was not likely extensive (Herbel et al. 2003). The majority of Se(−II) was found in the sediments as an Fe-Se precipitate, rather than in the solution phase as HSe^{-}. The resulting org-Se byproduct of the reductive process can be in gaseous, soluble, and solid phases. Selenium may substitute for S in organic compounds due to the similar chemical properties of these two elements.

Selenium has high affinity for organic phases in sediments. Selenium incorporated into the organic phase as extracted with NaOH was on average 33–49 % of the total sediment Se (Gao et al. 2007b). Further examinations showed that over 50 % of the Se in the NaOH extracts was Se(IV). It appeared that Se(VI) in drainage water was reduced and subsequently immobilized when the Se(VI) or the reduced species was incorporated into the organic phase.

Algae and bacteria indigenous to evaporation ponds were active in Se volatilization reactions (Frankenberger and Karlson 1989; Fan et al. 1997). The plant or microbial processes that form less toxic, gaseous Se species can result in the removal of Se from the drainage water in evaporation ponds via volatilization (Frankenberger and Karlson 1989; Terry and Zayed 1994). The resulting volatile Se compounds included dimethylselenide (DMSe), dimenthyldiselenide (DMDSe) and dimethylselenone (DMSe) (Thompson-Eagle and Frankenberger 1990a, b). The processes involved essentially an initial reductive reaction of Se(VI) to Se(IV) and/or Se(−II), followed by methylation of the reduced Se species to organic compounds or precursors of volatile DMSe (Cooke and Bruland 1987). The Se biomethylation process was primarily protein-peptide limited and thus could be promoted with the addition of a methyl donor, methionine, or a protein source (Thompson-Eagle and Frankenberger 1990a, b). Under natural environments, however, Se volatilization was found not to be extensive. Although 10–30 % of the total Se loss in a wetland system was reported from a Se(IV)-dominated wastewater (Hansen et al. 1998), a mass balance in a wetland system with vascular plants showed that Se volatilization losses only accounted for about 2 % of the total Se loss. Without addition of amendments, pond waters incubated for 43 days resulted in Se losses as DMSe, accounting for $\leq$6 % of total Se losses in water containing 0.7–1.7 mg Se L^{-1}. However, a much higher loss (35 %) was observed in a pond whose water had a much lower Se concentration, 0.014 mg L^{-1} (Thompson-Eagle and Frankenberger 1990a).

In summary, there is evidence of Se transformation as drainage water passes through the evaporation ponds, both as a result of volatilization and, more predominantly, by reduction, precipitation, or incorporation into the organic phase. These transformations result in a net reduction of soluble forms of Se in the water column. To alleviate risk to water birds, provisions were implemented at the evaporation ponds to discourage habitation by shore birds and, nearby, compensation wetland habitats were established with water of acceptable quality as alternative habitats (Tanji et al. 2003).

8.1.3.3 Arsenic

Arsenic (As) is a concern in evaporation ponds because, through the evapo-concentration process, As may be concentrated to potentially harmful levels

(Gao et al. 2007a; Ryu et al. 2010). Arsenic concentrations of drainage waters in Tulare basin are much higher than in other parts of the San Joaquin Valley because of the alluvial-lacustrine origin of basin-fill deposits. The behavior of As in evaporation ponds was similar to its behavior in the Owens Dry Lake bed where it accumulated up to 80 mg L^{-1} in shallow groundwater due to an intense evapoconcentration process and a highly reducing environment (Ryu et al. 2002).

Total As concentrations increased dramatically along the flow path as the ECF increased (Fig. 8.1, Gao et al. 2007a). In contrast to Se, As was conserved as it flowed through the evaporation ponds, although sink mechanisms were identified in some studies, as evidenced by poor correlations between As concentrations and evaporation or measured concentrations lower than the predicted values based on the ECF (Tanji 1990a; Ong et al. 1995; Ryu et al. 2010). Arsenic has four oxidation states (V, III, 0, and -III). The oxidized form, arsenate, As(V), is the dominant form under oxidizing conditions in drainage water. Arsenate is more strongly adsorbed onto soil or mineral surfaces than the reduced form arsenite, As(III), which is more soluble and more toxic. Elemental As is unstable and rarely exists in nature. Biomethylation results in a group of methylated compounds (org-As), such as monomethylarsonic acid (MMAA) and dimethylarsenic acid (DMAA), which were reported to be less toxic than As(III) (Cullen and Reimer 1989). Methylation is the major pathway resulting in As volatilization via formation of volatile arsines, including arsine, monomethyl arsine, dimethyl arsine, and trimethyl arsine (Anderson and Bruland 1991). Although no direct measurements have been made in evaporation ponds, As volatilization appears not to be a significant removal mechanism from pond water columns based on the linear increase in As concentration in pond waters with evaporation. Recent investigations identified all As species, i.e., As(V), As(III), and org-As (mainly as MMAA and DMAA) in evaporation pond waters with their dominance depending on redox conditions (Ryu et al. 2010).

As the ECF increased in pond waters, an increase in the concentration of total As and in reduced As species (As(III) and org-As) along the water flow path was recently reported (Gao et al. 2007a; Ryu et al. 2010). The fresh drainage water at the inlet and water in the initial drainage-receiving cell contained >95 % As(V), <5 % As(III), and negligible org-As. Arsenite and org-As reached 35 and 14% of the total As, respectively, towards the terminal cells. Concentrations of reduced As species were higher in the bottom of the pond water (near sediments) than in samples taken near the surface. The distribution of As species between cells and water depths corresponded well to the redox conditions, as indicated by DO, dissolved Fe, Mn, and N species, as well as by DOC. Arsenate remained dominant in the pond waters, accounting for >65 % of soluble As.

Sink mechanisms for removal of As from pond waters were observed (Tanji 1990a; Ong et al. 1995) and were investigated further by fractionating As deposited in the sediment (Ryu et al. 2010). Arsenic accumulation in sediment profiles showed that up to 80 mg As kg^{-1} was measured in surface sediments of terminal cells. In soil cores taken near the shorelines, no clear pattern of As distribution was observed with depth. The results showed the disturbances caused by an uneven sediment deposition due to bank erosion, wind-driven wave action, and/or buried sediment or detrital materials over time. Studies of the same pond systems at earlier dates, however, showed that the highest As concentrations were found in surface sediments, and they decreased with depth underneath the evaporation ponds (Ong et al. 1995). Similar observations were made in vegetated wetland sediments (Fox and Doner 2003). Chemical equilibrium modeling indicated that orpiment (As_2S_3) was most likely to form at high As and sulfide concentrations in saline and alkaline conditions (Ryu et al. 2002).

Ryu et al. (2010) examined As in sediment, using sequential extractions of KCl, K_2HPO_4 (pH 8), NaOAc (pH 5), NaOCl (pH 9.5), and NH_2OH-HNO_3, which obtained in sequence the As to represent insoluble, adsorbed, carbonate-associated, organic matter (OM)-associated, and in oxide forms, respectively. The fractionation was performed in surface sediments collected from two cells: the initial drainage-receiving cell, a more oxidizing environment with low EC and a near-terminal cell, a reducing environment with high EC. Results indicate that the reducing sediments contained more soluble and exchangeable As. Arsenic in association with oxides was appreciable only under oxidizing conditions. Carbonate minerals played an important role in immobilizing As into the sediments under alkaline conditions and in a broad range of redox potentials.

Sink mechanisms were significant for immobilizing As into sediments, resulting in little As being found in many pond waters in the alluvial fan region (Tanji 1990a; Ong et al. 1995). These mechanisms, however, were not significant for removing As from pond waters, as reflected by the increasing As concentration with ECF for evaporation ponds located in the Tulare Lake Bed (Fujii 1988; Gao et al. 2007a). Estimates of the mass ratio of total As in the 1 m depth of the surface water column in comparison to the 0.5 m depth of sediment resulted in an increase from 0.25 for the initial cell to 2.7 in the terminus cell, indicating that an increasing proportion of As occurred in evapoconcentrated waters, relative to the sediment, as water flowed through the cells (Gao et al. 2007a). Although not examined, soluble arsenic-thio compounds or arsenic-sulfide complexes were most likely present in evaporation pond waters, especially towards the terminal cells, as these compounds were detected in reducing environments, particularly in the presence of sulfide (Rochette et al. 2000; Wilkin et al. 2003).

Given the conservation of As through the sequence of evaporation ponds, it is apparent that natural mechanisms for the removal of As from agricultural drainage water may play a lesser role in any required remediation, particularly in the Tulare Lake Basin with its higher initial As concentrations in soils. The conservation and concentration of As from the inlet to the terminal drainage pond suggest that there is limited plant or bacterial volatilization occurring. Natural sink mechanisms and sequestration in sediments may not be adequate to reduce As concentrations when initial concentrations are high.

8.1.3.4 Other Trace Elements

Boron (B) is present as an uncharged ion in the aqueous solution, $B(OH)_3{}^0$ or $H_3BO_3{}^0$, at a wide pH range (pKa = 9.1). As B is not subject to redox transformations, no significant sink mechanism was identified. Despite some adsorption to metal oxides under oxidizing conditions, B is considered a conservative element, as indicated by its linear relationship to EC or the Cl^- concentrations in pond waters (Fig. 8.1).

Molybdate (MoO_4^{2-} or Mo(VI)) is the most oxidized form of molybdenum (Mo). It is soluble and prevalent in agricultural drainage waters. Adsorption of Mo to metal oxides, clay minerals, or $CaCO_3$ may occur at low pH but it may not be an important mechanism in evaporation ponds because little Mo(VI) adsorption occurs when pH >8 (Goldberg et al. 1996). Incubating sediments under reduced conditions caused substantial loss of Mo from solution, possibly due to precipitation of molybdenite (MoS_2) following reduction of Mo (VI) to Mo(II) and sulfate to S (−II); then, this precipitate could be oxidized or resolubilized when the sediment was exposed to oxidizing conditions (Amrhein et al. 1993). Accumulation of Mo in sediments was detected in a 3-year old wetland system and most of the accumulated Mo became water soluble upon drying of the sediment samples (Fox and Doner 2002, 2003).

The **uranium (U)** concentration in evaporation pond water may reach 1,000 μg L^{-1}, much higher than that of saline lakes and ocean waters (Westcot et al. 1993). Oxidized U(VI) was expected to be the dominant species in pond waters. Following incubation, the highly reducing and sulfidic pond sediment specimens contained 25 % U(VI) and 75 % U(IV), based on the X-ray near-edge absorption spectroscopy (Duff et al. 1997b). The soluble uranyl ions (U(VI) or (UO_2^{2+})) are unstable at alkaline pH, undergo hydrolysis, and form complexes with $(OH)^-$ and carbonate (CO_3^{2-}) ions, such as $(UO_2)_2CO_3(OH)_3^-$, $UO_2CO_3^0$, $UO_2(CO_3)_2^{2-}$ and $UO_2(CO_3)_3^{4-}$ (Duff et al. 1999). Ponds with high salinities and high alkalinities contained the highest aqueous U concentrations. The U(IV) species is sparingly soluble in aqueous systems. Under reducing conditions, U(IV) could also form soluble complexes with carbonates, as $U(CO_3)_4^{4-}$, and $U(CO_3)_5^{6-}$ and soil organic matter (Duff et al. 1997a). Lab incubation studies indicated that sediment U concentrations were highly correlated with total organic carbon, suggesting that U is bound to soil organic matter (Duff et al. 1997a). Only under strongly reducing conditions with organic matter amendments to pond sediments was most of the U(VI) reduced to U(IV) (Duff et al. 1999). Reduction of U(VI) to U(IV) occurred in the evaporation ponds and immobilized parts of the U into sediments, especially for those near the reducing surface sediments. However, the reactions may not be extensive, as significant accumulation of U in pond waters was found.

Vanadium (V) concentrations in agricultural drainage water and evaporation basin waters were elevated, compared to seawater, but were in the same range as the western salt-sink lakes (Westcot et al. 1993). Vanadium in nature can exist in oxidation states of +3, +4, and +5. Under oxidizing conditions, such as in agricultural drainage water, vanadate (VO_4^{2-}), the most soluble form, is dominant, which can be reduced to vanadyl (VO^{2+}) in reducing environments. Vanadyl ion forms strong complexes with organic matter. The soils from the Tulare Lake bed, when incubated at low redox potentials, resulted in a dramatic decrease of the V concentrations in soil suspension (Amrhein et al. 1998); similar studies of soils obtained from Kesterson Reservoir, however, indicated that V became more soluble (Amrhein et al. 1993). Aeration of reduced sediments led to re-solubilization of V (Amrhein et al. 1998). In constructed wetlands flooded with agricultural drainage water, the V accumulation in sediments appears to be affected by redox potential levels (Fox and Doner 2002, 2003). The V(III) redox state, occurring as precipitates of oxides, is expected to form under extremely reducing conditions (Fox and Doner 2002). The V concentrations in bottom sediment were 40 mg kg^{-1} in the drainage water-receiving cell and gradually decreased to 21 mg kg^{-1} in the terminus cell, reflecting the oxidation and reduction processes taking place, as the water flowed through the ponds, which may indicate more accumulation in initial drainage receiving cells (Fujii 1988).

8.1.4 Sink Effects of Evaporation Ponds

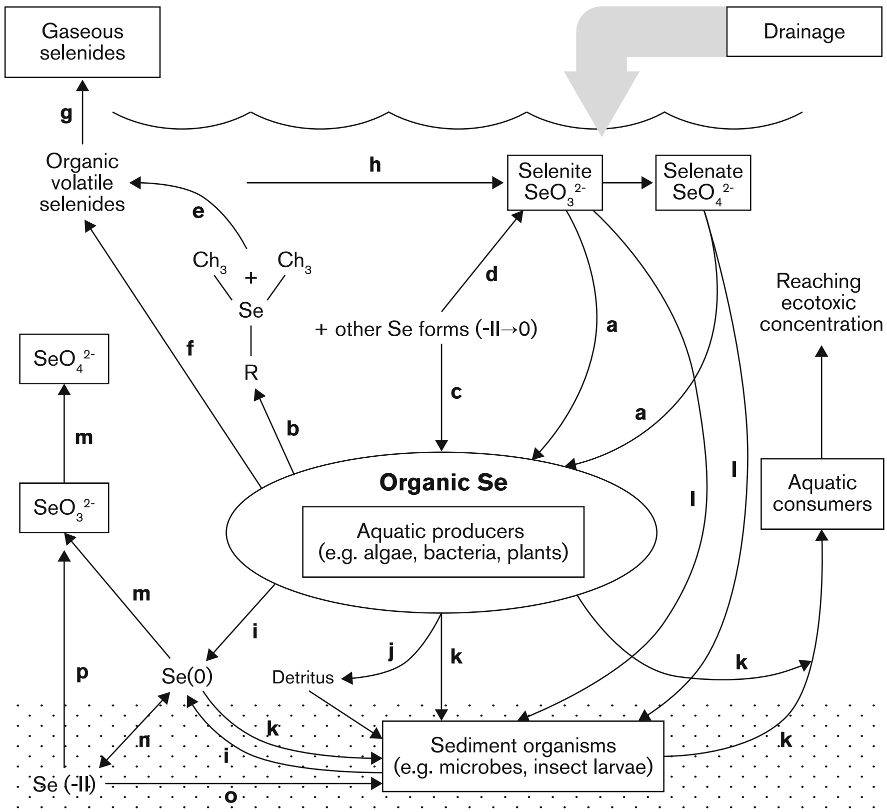
Sediments and soils underneath the evaporation ponds were identified as a sink for trace elements in the drainage water by Chilcott et al. (1993), because their concentrations in sediment were invariably higher than in surrounding soils. The surface 6 cm of the sediment layer was organically rich, exhibited reduced conditions, and contained elevated levels of B, As, Se, Mo, and U. At depths greater than 10–12 cm, concentrations of all trace elements, except for As, returned to the background levels of the soil. Similarly, the As, Mo, and V concentrations of sediments in reconstructed wetlands receiving agricultural drainage water were the highest in the top 2–4 cm of sediment layer and then gradually decreased with depth (Fox and Doner 2003). The capacity of soil, calcite, and goethite-coated quartz to retain As, Mo, and V in the reducing environment of sediments was determined by an *in situ* method whereby permeable bags containing the common minerals formed in evaporation ponds were placed at the sediment layer of a flow-through constructed wetland for 3 or 12 months (Fox and Doner 2002). When retrieved, As, Mo, and V were found on the surface of goethite-coated sand. Obviously, adsorption by goethite minerals would be a significant process to immobilize trace elements in the evaporation ponds. Meanwhile, calcite adsorbed slight amounts of Mo and V but little As, indicating this mineral is not an important sink for trace elements. The As and V concentrations of the soil, recovered after 12 months of exposure, changed little. The Mo concentrations under comparable conditions were 27 mg Mo kg^{-1} and 11 mg Mo kg^{-1} for soil recovered at water depths of 15 and 3 cm, respectively. However, nearly half of the Mo accumulated in soil became water-soluble after they were air-drying. The redox transformations play an important role in immobilizing trace elements into sediments, and when redox status changes in favor of oxidation, immobilized trace elements can be released. The potential for trace minerals immobilized in the pond sediments to become re-mobilized needs to be taken into account when bioremediation is employed to reduce the concentration of trace elements in the pond water.

8.2 Aquatic Food Chain and Biotransformation of Selenium

Bioaccumulation of Se in agricultural drainage water through the aquatic food chain initially triggered the eco-toxicological crisis at the Kesterson Reservoir in 1983. To prevent spread of the crisis, Kesterson Reservoir was capped promptly. At the time, there had not been adequate opportunities to document the entire processes, *in situ*. Since then, however, there has been quite a substantial improvement in understanding the pathways for mobilization, transport, transformation, and accumulation of Se in the irrigation – drainage-based crop production systems of the San Joaquin Valley (Dugan and Frankenberger 1999; Frankenberger and Karlson 1994; Karlson and Frankenberger 1990; Frankenberger and Arshad 2001; Higashi et al. 2005; Benson et al. 1999). There also have been laboratory and field studies of the biogeochemical processes of Se. The evaporation ponds, consequently, are microcosms in which processes and reactions of Se bioaccumulation in aquatic ecosystems may be observed and extrapolated to similar environmental settings. If birds are not protected from Se-laden surface water bodies, such as evaporation ponds, a Kesterson-like eco-toxicological episode may occur again.

Given the complexity of biogeochemical pathways in aquatic environments, addressing bioaccumulation through the food chain is a challenging task. Although Se present in the water itself is not the culprit, it undergoes multifaceted, dynamic and interactive biochemical transformations in water columns and in the underlying sediments (Fig. 8.2). While Se is a required nutrient for aquatic biota, the requirement is only marginally lower than the level that would result in eco-toxicological consequences (toxicity) to biota in upper echelons of the food chain. Impacts can be magnified along the food chain, even with relatively minor shifts in the biogeochemical equilibrium of Se in pond/sediment ecosystems. Fan et al. (1998) document that Se volatilization may be both a means of removing waterborne Se from the system and an important link to either ecotoxic risk or to mitigation of Se-contaminated water, depending on the pathways that define the chemical forms of Se available in the water and food chain. The filamentous cyanophyte-dominated mat in the evaporation basins previously referenced in this chapter was active in converting SeO_3^{2-} (selenite) into gaseous org-Se. Selenite might also be incorporated into proteins, primarily in the form of selenomethionine, thus the advent of food chain transfer of Se in the aquatic ecosystem.

Stewart et al. (2004) investigated the food web pathways for Se bioaccumulation in San Francisco Bay and noted the dominant bivalve *Potamocorbula amurensis* had a tenfold slower rate constant for eliminating the Se body burden than common crustaceans, such as copepods and the mysid *Neomysis mercedis*, wherein the Se elimination rate constants were $k_e = 0.025$, 0.155, and 0.25 day^{-1}, respectively. Analyses based on stable isotope ratios showed that this difference would propagate up the respective food webs in San Francisco Bay. Concentrations of Se in tissues of zooplankton predators were lower than those of predator bivalves, which had Se tissue concentrations that exceeded thresholds thought to be associated with



Legend

a. Uptake and transformation of Se oxyanions by primary and secondary producers
b. Release of selenonium and other organic Se metabolites by aquatic producers
c. Uptake of organic Se compounds by aquatic producers
d. Abiotic oxidation of organic Se compounds to Se oxyanions
e. Release of alkylselenides from selenonium or other alkylated Se precursors via abiotic reaction
f. Release of alkylselenides from selenonium or other alkylated Se precursors via aquatic producers
g. Volatilization of alkylselenides into the atmosphere
h. Oxidation of alkylselenides to Se oxyanions
i. Formation of red amorphous elemental Se [Se(0)] by aquatic and sediment producers
j. Detrital formation from aquatic producers
k. Se bioaccumulation into the food chain
l. Assimilation of waterborne selenium oxyanions into sediment biota
m. Oxidation of sediment Se(0) to oxyanions
n. Reduction of sediment Se(0) to Se(-II) or the reverse
o. Assimilation of sediment Se(-II) into sediment biota
p. Oxidation of sediment Se(-II) to selenite

Fig. 8.2 Biogeochemical cycling of Se in aquatic ecosystems (Redrawn from Higashi et al. 2005) teratogenesis or reproductive failure when the liver Se concentration was >15 μg Se g^{-1} dry weight. Stewart et al. (2004) concluded that basic physiological and ecological processes could lead to wide differences in exposure and physiological effects among species.

For organisms at the top of the aquatic ecological echelon, such as fishes and foraging birds, exposures to Se are essentially through the dietary intake. The schematic depictions of Se transfer in aquatic environments (Fig. 8.2) illustrate the dynamic and interactive pathways through which inorganic Se species may be incorporated into aquatic organisms and eventually lead up the food chain. Higashi et al. (2005) noted the variability of Se accumulation in algae, plants, and aquatic and benthic invertebrates in terms of bioconcentration factors, indicating that different ecological pathways of aquatic transformations might have led to dissimilar Se body burdens in the recipient organisms (Stewart et al. 2004). Clearly, the inorganic Se concentration of the water column would not be an appropriate indicator of the aquatic toxicity threshold.

8.2.1 Food Chain Pathways in Se-Contaminated Waters

Recognizing that Se bioaccumulates in aquatic ecosystems via the food chain, Schlekat et al. (2004) focused on biota at the lower trophic levels. Fan et al. (1997) reported that microalgae (*Chlorella sp.*) isolated from a saline evaporation pond were active in volatilization of alkylselenides, production of putative selenonium precursors of alkylselenides, and precipitation of Se and did not exhibit high accumulation of toxic selenomethionine in free form. Fan et al. (1998) further illustrated the metabolic pathways of Se volatilization and its accumulation in the filamentous cyanophyte-dominated mat found in agricultural drainage water evaporation ponds, which involved multiple Se transformations.

Dissolved Se speciation influences Se uptake by primary producers (such as phytoplankton) and microbes (Presser and Luoma 2006) and for these lower trophic organisms, uptake of Se(IV) in SeO_3^{2-} (selenite) is substantially more efficient than uptake of Se(VI) in SeO_4^{2-} (selenate). As the SeO_4^{2-} concentrations in the water column are considerably higher than that of SeO_3^{2-}, the actual Se uptake via these two inorganic Se species could be equally important. The bioconcentration factors by phytoplankton for SeO_3^{2-} can be as high as 10^5 (Fowler and Benayoun 1976; Wrench and Measures 1982). Once taken up, SeO_3^{2-} is incorporated into selenoamino acids within phytoplankton (Wrench 1978; Wrench and Campbell 1981), which are then transferred to the next trophic level with great efficiency. Assimilation efficiencies for phytoplankton-associated Se range from 55 to 90 % among different invertebrates (Reinfelder et al. 1997).

Se transformations by freshwater microalgae (*Chlorella* sp.) are influenced by the S and nitrate contents of the water. Neumann et al. (2003) incubated microalgae without nutrients and their subsequent transfer to a SeO_4^{2-} (selenate) solution resulted in rapid (24-h) volatilization and accumulation of Se; this response was inhibited by the addition of 1 mM sulfate but not by addition of nitrate. Reduction precursors were selenomethionine and selenocysteine. High rates of Se reduction and volatilization in the sulfate-deprived microalgae were attributed to reduced competition with the chemically analogous sulfate ions, favoring selenate uptake via up-regulated sulfate/selenate transporters, leading to a rapid reductive metabolism of selenate. In an experiment using both strains of the green alga *Scenedesmus quadricauda*, which were resistant to specific Se species, and using a wild alga strain, Umysova et al. (2009) also found a metabolic link between Se concentration, S availability, and toxicity. All strains experienced increasing toxicity related to dose and S deficiency. The total amount of Se and selenomethionine in biomass increased with the concentration of Se in the culturing media. For the wild strain and strains not resistant to the Se species used in the culturing media, thioredoxin reductase activity increased with Se concentration, which was hypothesized to indicate a stress response to Se cytotoxicity.

Although not directly related to Se contamination in agricultural drainage water evaporation ponds, several studies of the San Francisco Bay reinforce the conclusion that Se bioaccumulation pathways are complex and vary substantially by species. There was early confirmation that uptake of dissolved Se (dissolved SeO_3^2 [selenite] plus dissolved org-Se) by invertebrates is not as important as uptake from diet (Luoma et al. 1992; Lemly et al. 1993). At the same time, Luoma et al. (1992) showed that the uptake rate of dissolved SeO_3^2 explained <5 % of the Se tissue burden accumulated by the clam *Macoma balthica* at concentrations typical of the Bay-Delta. For this species, the role of dissolved organic selenides in Se bioaccumulation is not as well understood as the availability of inorganic Se, but it is unlikely that the rate of uptake is sufficient to be greater than uptake rates from food. An example of the influence of confounding processes among these links can be found in data from the Bay-Delta watershed (Presser and Luoma 2006). Black-necked stilts (wading birds) exhibited similar levels of Se accumulation, with 20–30 μg Se g^{-1} dry weight found in eggs at Chevron Marsh in the Bay-Delta and 25–37 μg Se g^{-1} dry weight found in eggs at Kesterson Reservoir, but the source water in Chevron Marsh contained only about 20 μg Se L^{-1}, 10 % of the Se concentration found at Kesterson Reservoir where the maximum concentrations were 300 μg Se L^{-1} (Skorupa 1998). The reason for this counterintuitive finding was that Se at the Chevron site was exclusively SeO_3^2 (Presser and Luoma 2006). Because of such complexities, the strongest correlative predictor of Se concentrations in predator tissues, based on Se exposures, is probably the Se concentration in invertebrates (Presser and Luoma 2006). However, there have been few investigations to explore feeding relationships and resultant correlations with Se bioaccumulation in food webs.

8.2.2 Transformations and Bioremediation

There are three basic strategies for remediation of toxins in the environment: containment, removal, and treatment (Brusseau and Miller 1996). Containment is the fundamental strategy associated with fixed standards for discharges/releases of contaminants to the environment. If a discharge/release standard is exceeded, then the discharge is prohibited and the discharging party is tasked with meeting the standard before releasing to the environment again. Removal and treatment are post-release remediation strategies. All three of these strategies are difficult to apply to Se because (a) initial Se concentrations in the "contaminated" water released to the environment are typically in the parts-per-billion (ppb) range; (b) Se is chemically similar to S, which can be present at concentrations many orders of magnitude greater than those of Se; and (c) the releases of irrigation drainage water containing Se are measured in tens of thousands of cubic meters (Higashi et al. 2005).

Containment is difficult because Se contamination (a) occurs in open lentic systems, such as reservoirs and evaporation ponds, in lotic systems, such as streams and rivers, and in receiving waters, such as deltas and estuaries (Higashi et al. 2005); and (b) there are numerous sources of Se in the environment, including drainage water from thousands of hectares of agricultural land, industrial wastewater from oil refining, and soils with high Se levels (Presser and Luoma 2006). Simple containment of lotic Se in terminal evaporation basins avoids release of water into lotic systems, but it does not address the food chain accumulation problem in either lotic or lentic ecosystems (Tanji et al. 2003).

Simple removal of Se is also a complex and, to some extent, reversible process (Fig. 8.3), and may or may not address the issue of bioaccumulation, particularly if the remediation target is a regulatory TMDL (Higashi et al. 2005). As noted in earlier chapters, there are a number of bacterial, algal, and plant-mediated pathways for Se removal that may be feasible (Tanji et al. 2003; Frankenberger and Arshad 2001). Frankenberger and Karlson (1994) presented a volatilization scheme (Fig. 7.1) that has been tested in the field but requires additional research to determine the efficiency of upscaling to become a practical regional solution (Frankenberger and Arshad 2001). The various pathways for organismal transformation of bioavailable Se shown in Fig. 7.1 (in this case, soil fungi) need to be characterized fully to identify the mechanisms by which various management factors affect the rates and sustainability of such transformations.

Treatment options, such as those involving a combined removal-treatment strategy proposed by Gerhardt et al. (1991), are focused on transformation of bioavailable Se to elemental or other stable mineral forms of Se via bacterial, algal, or plant reduction. To the extent that these methods are focused on reduction of total Se in the water column and not on the bioaccumulation issue *per se*, they may accomplish the treatment objective without addressing the end problem. Gerhardt et al. (1991) used the dissimilatory reduction capability of bacteria to reduce SeO_4^{2-} to Se(0) and upscaled the method to the field trial scale. The method was shown to remove approximately 80 % of the total Se from the water column, but the combined microbial and algal action increased the levels of org-Se and selenite by 2–4 times (Amweg et al. 2003). This result illustrates again the problem of a regulatory focus and a traditional TMDL standard for a toxin such as Se that rapidly bioaccumulates at lower trophic levels and moves up

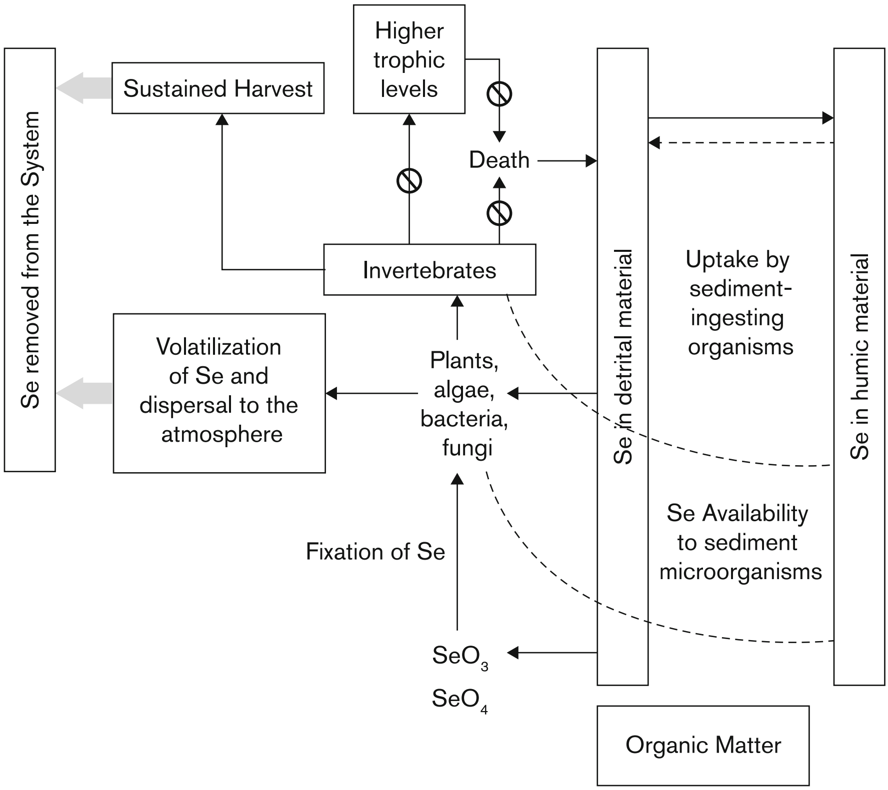


Fig. 8.3 Bioremediation pathways for breaking food chain transfer of Se resulting in toxicity at higher trophic levels

the food chain. Given the complexity of the biochemical pathways of Se transformation in a myriad of species, there are inherent risks in containment, removal, and treatment strategies (Higashi et al. 2005). In addition, bioremediation is a moving target. Lemly (1997) noted, "The predominant cycling pathways that exist during periods of selenium input may shift dramatically once inputs stop. For example, waterborne routes of exposure to fish and wildlife may be replaced by sediment-detrital pathways."

8.3 Integrated Management as an Alternative

Given the difficulty in establishing a universal, protective standard for Se in the environment and given the inherent complexities of the biochemical pathways of various forms of Se bioremediation, it may be appropriate, at least in the near-term, to redefine the objective of Se bioremediation. In metaphoric terms, an alternative to untying the Gordian knot may be to cut it. This involves re-thinking the objectives of remediation. The underlying problem at Kesterson Reservoir was the bioconcentration of Se in a system used by wildlife. An alternative strategy for addressing the risk to wildlife at the upper trophic levels is to manage Se and simultaneously interrupt its flow along the food chain, thereby reducing toxicity to biota at the higher trophic levels. To break Se flow along the food chain requires the use of an organism that bioaccumulates Se and is readily harvestable so that the organism itself does not accumulate on site.

This approach has been the subject of a large-scale field experiment (Higashi et al. 2003; Rofen 2001). As described in Higashi et al. (2005), this experiment in irrigated drainage evaporation ponds at Tulare Lake Drainage District (TLDD) combines bio-volatilization of Se by the primary producers in the aquatic ecosystem and the use of brine shrimp, herbivores occupying the bottom echelon of the grazing food chain, as an accumulator of Se. When the brine shrimps are continuously harvested, the flow of Se along the aquatic food chain is interrupted and the mass of accumulated Se is separated from the evaporation ponds. Invertebrates occupy a critical middle link and aid in trophic transfer of Se in many terrestrial and freshwater food chains, and there are numerous studies that attribute biomagnifications of Se in invertebrates as a link in the food chain to higher trophic organisms (Morrissey et al. 2005; May et al. 2001). Selenium is known to biomagnify as it moves from periphyton to plankton, to detritivore and predator invertebrates (Muscatello et al. 2008).

There were four components of the TLDD field experiment: (a) algae that volatilize Se grow naturally in the evaporation basins; (b) algae are the food base for brine shrimp that grow naturally and in high densities in the basins; (c) water management techniques in the experiment were typical of those already in practice; and (d) brine shrimp can be harvested and marketed to partly offset the costs of management. In the process, the invertebrate biomass is decreased substantially, and removal of brine shrimp precludes the return of Se to the detritus because the shrimp accumulate the Se in their tissues. There is a net 90 % reduction in Se deposited to the sediment, and a corresponding reduction in the Se available for refluxing (see Fig. 8.3).

The application of a multifaceted, integrated strategy for management of both Se and of the food chain, rather than a focus solely on Se containment, remediation, or treatment alone, avoids the problem of addressing the very complex nature of Se contamination and provides a practical solution to key elements, but not all aspects, of a multifaceted environmental problem. When compared to management of a suite of environmental conditions to directly address the issue of Se contamination (nitrate levels, carbon sources, sulfate competition with Se, temperature, oxygen levels, and so forth), the TLDD management intervention is an elegantly simple one: harvest the invertebrate link to the upper trophic level animals to remove Se from the system and effectively break the food chain. The remaining functions that directly affect Se loading and speciation occur naturally, and may not need to occur at optimum rates for the management to succeed.

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Chapter 9
Management of Evaporation Basins To Reduce and Avoid Adverse Impacts to Waterbirds

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The Tulare Lake Drainage District (TLDD) manages the collection, conveyance, and disposal of subsurface agricultural drainage water for a primarily agricultural area of 89,529 ha (221,230 acres), of which 11,356 ha (28,061 acres) are equipped with tile drain collection and conveyance facilities. The collected drainage water is conveyed to three evaporation basins, with a total wetted surface area of 1,280 ha (3,165 acres). Maximum water storage capacity within the evaporation basins totals 13.3 $\times$ 10^6 m^3 (10,795 acre-foot). Annual evaporation at the three basins averages 16.6 $\times$ 10^6 m^3 (13,436 acre-foot). The evaporation basins are operated as a closed hydrologic unit having no surface water discharge. Evaporation basins have been in operation in the Central Valley since the mid-to-late 1970s. Without the evaporation basins for agricultural drainage water disposal, agricultural productivity within the area would be substantially diminished or eliminated. In this context, the only option for sustainable irrigated agriculture in the Tulare Lake Basin is to manage the evaporation basins in a manner that avoids and minimizes the risk of adverse impacts as a result of exposure to waterborne selenium, salts, and other stressors. And, to the extent that this is not feasible, to mitigate the impacts to wildlife that occur.

Various species of shorebirds and waterfowl are attracted to the evaporation basins, actively foraging and nesting there, and are therefore exposed to naturally occurring trace elements in drainage water, which include arsenic, boron, molybdenum, selenium, uranium, and vanadium. The trace element of most concern in Central California evaporation basins is selenium (Se), which is toxic to shorebirds and waterfowl when concentrations are elevated (Ohlendorf et al. 1993; Ohlendorf 2003). Other trace element constituents do not appear to cause reproductive

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_9,

failure and mortality at concentrations found in evaporation basins (Ohlendorf et al. 1993). Exposure to a mix of trace elements could result in (1) increased mortality, (2) reduced growth or impaired condition, (3) reproductive impairment, (4) reductions in species abundance, or (5) cumulative effects, when viewed with results from past, current, and future projects (Hoffman and Heinz 1988; Ohlendorf et al. 1988; Paveglio et al. 1992, 1997; Skorupa and Ohlendorf 1991). One of the most significant environmental risks is associated with reproductive impairment (Skorupa and Ohlendorf 1991), including the risk of reduced hatchability and teratogenesis for American avocets (*Recurvirostra americana*) and black-necked stilts (*Himantopus mexicanus*). These two species were the most numerous shorebirds using the evaporation basins, with annual use by several thousand individuals recorded in the 1980s and early 1990s.

Based on results of water quality monitoring, waterborne Se concentrations at two of the TLDD evaporation basins were found to meet or exceed threshold levels for adverse effects to shorebirds. In addition to concerns regarding shorebird foraging and nesting at the evaporation basins during the spring and early summer, concern was also expressed regarding the potential exposure of wintering waterbirds to hypersaline conditions that could lead to salt deposition on feathers and other adverse effects. Faced with this conflict between ongoing agriculture and a suite of protected shorebirds and waterfowl, beginning in the late 1980s, the TLDD, the Regional Water Quality Control Board (RWQCB), the California Department of Fish and Wildlife (CDFW), the U.S. Fish and Wildlife Service (USFWS), as well as the University of California (UC) Salinity Drainage Program began a series of investigations to improve understanding of the following issues:

- Potential risks that evaporation basin operations posed to wildlife,
- Chemical constituents of agricultural drainwater from various geographic locations within the San Joaquin Valley (SJV) (site-specific studies),
- Chemical dynamics and the physical and biological processes affecting Se and other trace element constituents within evaporation basins,
- Alternative methods for disposal (e.g., use of saline drainwater to irrigate salt-tolerant crops),
- Use of wetland vegetation to assimilate and volatilize Se, and
- Identification and testing of various methods for reducing and avoiding potentially significant adverse impacts to wildlife.

For more than a decade, the TLDD, working collaboratively with the UC Salinity Drainage Research Program and state and federal resource and regulatory agencies, has been actively engaged in all facets of the above-referenced investigations. Much of the outcomes were summarized in Chap. 8, Chemical and biological processes of evaporation ponds. The TLDD has conducted biological monitoring and analyses (1) to assess the potential for adverse impacts associated with TLDD evaporation basin operations, (2) to identify and implement measures that reduce and avoid impacts to wildlife, and (3) to design and operate both a spring-summer wetland using low-Se saline drainage water for foraging and nesting by shorebirds (avocets and stilts), which compensates for unavoidable evaporation basin impacts, and a winter waterfowl wetland habitat, also supported by the use of low-Se saline drainage water supplies, to increase availability and diversity of winter waterfowl habitat within the area. A brief case history is presented below, based on the experiences, monitoring, analyses, and tests conducted at the TLDD facilities over the 10-year period from 1994 through 2004. Testing and performance monitoring of the existing measures implemented at the TLDD evaporation basins, as well as testing and experimentation with new technologies are continuing to explore alternatives and refinements in agricultural drainage water management and disposal within the TLDD service area and throughout the SJV.

Application of research findings for the 1994–2004 decade discussed below were managed quite differently from those associated with the Kesterson Reservoir crisis in the prior decade. The Se concentrations at Kesterson were typically greater than 600 μg Se L^{-1}. in contrast to Se concentrations at the TLDD evaporation basins of less than 20 μg Se L^{-1}.). The response to Se toxicity at Kesterson was to assume that the ponds could not be managed and must be abandoned; given the lower Se concentrations at the TLDD ponds, the response was to consider a suite of actions that, in concert, might address the toxicity issue from a variety of perspectives. The various actions fall into three general categories:

- Manage the Se in the ponds to reduce the potential impact on birds and other wildlife using them;
- Manage the ponds to reduce their attractiveness to birds and wildlife and thus limit their exposure to Se in the ponds; and
- Develop alternative habitats for the birds that would be more attractive and would safely provide the wetlands habitat needed.

In short, the problem addressed by the TLDD program was not defined as narrowly as it was at Kesterson Reservoir; the focus was on the interaction of the birds and the ponds, not just on the Se in the ponds. The research included work to identify the factors that might be manipulated to keep the birds out of the ponds, where they could encounter toxic levels of Se, as well as to identify ways to break the food chain that had resulted in bioaccumulation of Se at Kesterson. Thus, the research from 1994 to 2004 did not assume a priori a solution to the problem, or even a particular strategy for a solution. And, most importantly, the work at TLDD was not carried out in a crisis environment.

9.1 Assessment of Wildlife Impacts at the TLDD Evaporation Basins

9.1.1 Wildlife Use of the Ponds

Before assessing the impacts, it was essential to know which birds were using the ponds and when their use occurred. Wintering and migratory waterbirds use the evaporation basins for rafting, foraging, and nesting. Peak numbers of waterbirds were present at the basins during fall migration when shorebirds of various species mostly western and least sandpipers (*Calidris mauri* and *Calidris minutilla*), phalaropes (*Phalaropus tricolor*), long-billed dowitchers (*Limnodromus scolopaceus*), yellowlegs (*Tringa* spp.), and dunlin (*Calidris alpina*) use the basins. Grebes (*Podiceps nigricollis*) are also present at the evaporation basins during the winter months and throughout the year. Diving ducks (mostly ruddy ducks, *Oxyura jamaicensis*) are abundant during winter at the evaporation basins. During the spring and fall migration, dabbling ducks, diving ducks, and shorebirds are present in the highest densities.

Snowy plovers (*Charadrius alexandrinus*), American peregrine falcons (*Falco peregrinus*), long-billed curlews (*Numenius americanus*), California gulls (*Larus californicus*), white-faced ibis (*Plegadis chihi*), double crested cormorants (*Phalacrocorax auritus*), American white pelicans (*Pelecanus erythrorhynchos*), northern harriers (*Circus cyaneus*), Swainson's hawks (*Buteo swainsoni*), golden eagles (*Aquila chrysaetos*), prairie falcons (*Falco mexicanus*), black terns (*Chlidonias niger*), loggerhead shrikes (*Lanius ludovicianus*), tri-colored blackbirds (*Agelaius tricolor*), short-eared owls (*Asio flammeus*), and burrowing owls (*Athene cunicularia*) also have been found on evaporation basins in the Central Valley (U.S. Fish and Wildlife Service 1995). White-tailed kites (*Elanus leucurus*) are also assumed to occur at the basins. Other special-status species that may be found at the basins are sharp-shinned hawk (*Accipiter striatus*), Cooper's hawk (*Accipiter cooperii*), and ferruginous hawk (*Buteo regalis*). However, their use of the basins would probably be irregular and infrequent, generally being limited to birds flying over the site.

Few mammals have been documented using the basins. The most common mammals include coyotes (*Canis latrans*), raccoons (*Pocyon lotor*), and striped skunks (*Mephitis mephitis*), which travel along the perimeter and interior levees searching for food, especially bird eggs or young. Rodents, reptiles, and amphibians also occur in the area in low numbers.

A number of criteria have been used to evaluate the potential impact upon wildlife species using evaporation basins (Swanson 1988). Impact assessment included the following: (1) the numbers and species composition of wildlife, particularly waterbirds inhabiting the evaporation basins on a seasonal basis; (2) the number of birds nesting at the evaporation basins; (3) Se concentrations in (a) the water, (b) macroinvertibrates preyed upon by foraging birds, (c) bird tissue, and (d) particularly bird eggs; (4) the risk of adverse impacts on reproductive success for those birds foraging and nesting at the evaporation basins, based on population-level toxicological risk associated with dietary exposure and Se concentrations observed in waterbird eggs; (5) the observed frequency of reproductive impairment including, but not limited to, teratogenesis; and (6) risk of acute and chronic effects on wildlife associated with exposure to elevated concentrations of water quality constituents including, but not limited to, Se and salts. Agricultural drainage water constituents other than Se that may have adverse ecotoxic effects include arsenic (As), boron (B), molybdenum (Mo), and uranium (U).

When impacts were observed, in accordance with California Environmental Quality Act (CEQA) Guidelines, mitigation measures were identified which would accomplish the following objectives: (1) avoid adverse effects associated with evaporation basin operations, (2) reduce and minimize site-specific evaporation basin impacts, and/or (3) compensate for unavoidable site-specific impacts through design and operation of an off-site compensation wetland habitat. The evaporation basin operations comply with CEQA requirements if on-site actions designed to avoid and minimize adverse effects, in combination with off-site compensation for unavoidable losses, reduce adverse impacts to less-than-significant levels.

9.1.2 Selenium Effects: Efforts to Define Toxicity

Since the initial studies following the crisis at Kesterson, there has been an ongoing debate regarding the usefulness of a waterborne Se standard for toxicity and considerable new research has occurred to define the thresholds of Se toxicity in a variety of organisms. Because Se is an essential nutrient, bioavailable forms of Se are readily taken up by many species and species have adapted physiologically to low-Se and high-Se environments. The toxicity of Se would therefore be expected to vary by species.

Elevated levels of Se have been documented to have adverse effects on wildlife (CH2MHill et al. 1993; Skorupa and Ohlendorf 1991; Skorupa 1998; Adams et al. 2003; Ohlendorf 2003, and others). Most of the research has been conducted on waterbirds in whom Se effects range from subtle, sublethal changes, such as weight loss, to reduced hatchability (i.e., the portion of a clutch of eggs that develops to the hatching stage) and teratogenesis (i.e., developmental malformations of embryos). The levels of Se at which the probabilities of these effects are likely to increase (referred to as threshold levels) have been analyzed and calculated for breeding birds inhabiting the SJV (Skorupa and Ohlendorf 1991; CH2MHill et al. 1993; Maier and Knight 1994; Ohlendorf 2003; Adams et al. 2000; Parametrix 2002). These thresholds are reported as the amount of Se in the water (waterborne Se), food chain biota (dietary Se), and bird tissues (avian eggs, breasts, and livers) resulting in adverse affects (e.g., impaired egg hatchability).

One of the first efforts to evaluate Se toxicity thresholds for waterbirds within the SJV evaporation basins was a statistical linear regression model developed by Skorupa and Ohlendorf (1991). The model included two separate linear regressions designed to assess trophic pathways from water into the aquatic food chain and then consumed by waterbirds and ultimately transferred maternally to waterbird eggs. Selenium concentrations within the waterbird eggs were then used as a predictor of the potential for adverse impacts, such as an increased probability of embryonic abnormalities. The relationship developed between waterborne Se and food chain Se concentrations was based on field data collected from several evaporation basin sites within the SJV. The relationship between dietary Se and mean egg Se was based on experimental studies with farm-raised mallards exposed to a selenomethionine-spiked diet. For purposes of the model, food chain Se and dietary exposure to Se were assumed to be equal. Mean egg Se values were based on data for eared grebes, assuming a value of 3 parts per million (ppm) dry weight (DW) was equal to natural background concentrations. Using the regression models, Skorupa and Ohlendorf (1991) estimated waterborne Se concentrations of 0.5–2.3 μg Se L^{-1} as the range that would not result in egg Se levels of 3 ppm. Based upon results of these analyses, a waterborne Se criterion of 2 μg Se L^{-1} was recommended by the USFWS as a chronic Se water quality criterion for the protection of waterbirds. For purposes of evaluating the risk of reproductive impairment for avocet and stilts, Skorupa and Ohlendorf (1991) proposed an egg Se threshold of 6 ppm (DW), based on statistical analysis of an EC_3 threshold. Results of the Ohlendorf and Skorupa (1991) analysis were characterized by relatively high variability ($r^2 = 0.28$, Adams et al. 2003), such that the EC_3 of a 6 ppm egg Se concentration could not be distinguished reliably from naturally-occurring background levels (Adams et al. 2003). Ohlendorf and Santolo (1994) utilized the same regression model, but substituted a mean egg Se concentration of 8 ppm, which was based on the threshold concentration for impaired egg hatchability in American avocet and black-necked stilt populations in the Tulare Basin. Utilizing the 8 ppm mean egg Se value, a waterborne Se concentration threshold was estimated to be 7.8 μg Se L^{-1}.

Since publication of the original Se risk analyses by Skorupa and Ohlendorf (1991) and Ohlendorf and Santolo (1994), data from a wide variety of field and laboratory experiments have been compiled and have been subject to additional, rigorous statistical analysis to evaluate further the potential relationships between waterborne Se, dietary Se, and egg Se concentrations, resulting in various biological endpoints characterizing different levels of reproductive impairment (e.g., reduced egg hatchability, embryonic deformities and abnormalities, increased juvenile mortality and failure to thrive) for various species of waterbirds. Selenium has been found to affect embryo development adversely and, at sufficiently high concentrations, can result in gross abnormalities, which may include anophthalmia, incomplete or deformed beak development, brain defects, foot defects, and other gross embryological abnormalities. More subtle teratogenic effects, such as an enlarged heart, edema, liver hypoplasia, and gastroschisis, may also occur but are more difficult to detect. These subtle effects are more easily detected through reduced embryo and chick survival (e.g., reduced hatchability).

In the 1990s, after the Kesterson crisis had been resolved with the closure and capping of the reservoir ponds, more comprehensive investigations were performed to evaluate further Se toxicity thresholds for adverse impacts to reproducing waterbirds. These investigations included additional analysis of both laboratory and field data on Se concentrations in water, food, and waterbird eggs using sophisticated statistical analyses and modeling techniques to identify ecologically significant Se threshold concentrations resulting in reproductive risk. Many of these investigations have also focused on evaluating site-specific variability in Se risk thresholds for both fish and wildlife, reflecting unique environmental conditions that occur within a site, affecting Se pathways and their effects on Se toxicity and risk of adverse impacts (see, for example, Adams et al. 1998, 2000, 2003; DeFrost et al. 1999; Fairbrother et al. 1999; Ohlendorf 2003, and other investigators). Results of these investigations suggest the following:

Selenium bioaccumulation varies from site to site and in response to a variety of environmental variables, including the biogeochemistry of Se cycling processes, and the resulting concentrations of Se in organic and inorganic forms (e.g., bioavailable Se concentrations). As described in Chap. 7, Se transfer is highly variable, based upon a variety of site-specific factors (Higashi et al. 2005; Adams et al. 1998). Adams et al. (2000) identified site-specific differences between lotic (flowing) and lintic (standing) water bodies. Results of these analyses demonstrated significant differences in the bioaccumulation of Se by fish and invertebrates from flowing and standing water and found that selenate was much less bioaccumulative than selenite. Considerable variation in Se accumulation by waterbirds was also identified as a site-specific factor to be taken into account when establishing water quality criteria.

The risk of adverse toxicity is greater for organic forms of Se when compared to inorganic Se. The ratio of organic to inorganic Se varies from one site to another based on a variety of factors (e.g., redox, pH, biological productivity, oxidation, and biotransformation), which affect the risk of ecologically significant impacts, but are not necessarily reflected in the measurement of total waterborne Se concentrations.

Toxicity thresholds for reproductive problems vary substantially among waterbird species. The estimated egg Se concentrations (EC_{10}) resulting in teratogenesis for ducks, stilts, and avocets have been found to be significantly different, with egg threshold concentrations for mallards of 23 ppm, black-necked stilts of 37 ppm, and American avocet of 74 ppm (Ohlendorf 2003). Egg concentrations resulting in reduced hatchability have been found to be lower than those causing embryo deformities. Using the more sensitive, reduced hatchability criterion, Fairbrother et al. (1999, 2000) statistically re-analyzed available laboratory and field data regarding the relationship between egg Se and reduced hatchability for mallard ducks. Results of these analyses showed an egg Se threshold concentration for reduced hatchability (EC_{10}) of 16 ppm. Ohlendorf (2003) conducted further analyses of available data for mallard ducks and concluded that the geometric mean egg Se threshold concentration that resulted in reduced hatchability (EC_{10}) was 12.5 ppm. After a detailed statistical review of available field and laboratory data on the effects of Se on waterbird eggs (e.g., hatchability, mortality, teratogenesis), Adams et al. (2003) identified an egg Se effects threshold of 12–14 ppm (DW) as appropriate and conservative for assessing potential adverse environmental effects. This recommended egg threshold concentration is consistent with the results and recommendations by Ohlendorf (2003). Based on the range of threshold concentrations of waterbird eggs identified from these analyses, waterborne Se concentrations associated with egg Se concentrations ranging from 8 to 14 ppm (DW) were analyzed for use in identifying site-specific waterborne Se criteria for operation of the TLDD Compensation Habitat as well as for assessing potential impacts of evaporation basin operations.

Using a "hockey stick statistical method" to evaluate toxicity, as measured by reduced hatchability for black-necked stilt eggs, Adams et al. (2003) produced EC_{10}

Table 9.1 Summary examples of egg selenium threshold response analyses

Species/response	Reported egg Se thresholds (μg g^{-1} dry weight)	References
Stilt/clutch inviability	6[a]	Skorupa and Ohlendorf (1991)
Stilt/impaired hatchability	8	Ohlendorf and Santolo (1994)
Duck/teratogenesis	23[b]	Ohlendorf (2003), Skorupa (1998)
Stilt/teratogenesis	37[b]	Ohlendorf (2003), Skorupa (1998)
Avocet/teratogenesis	74[b]	Ohlendorf (2003)
Duckling/mortality (7 day)	16[b]	Fairbrother et al. (2000)
Duck/impaired hatchability	12.5[b]	Ohlendorf (2003)
Stilt/teratogenesis	15.5–24.3[b]	Adams et al. (2003)
Duck/teratogenesis	21[b]	Adams et al. (2003)
Duck/teratogenesis	23[b]	Adams et al. (2003)
Stilt/egg inviability	21–31[b]	Adams et al. (2003)
Stilt/egg inviability	24.2[b]	Adams et al. (2003)
Duckling/mortality (7 day)	14–15[b]	Adams et al. (2003)
Duck/egg inviability	12.3[b]	Adams et al. (2003)

Source: Adams et al. (2003), Ohlendorf (2003)
Results reflect statistical analyses of field and laboratory-derived data using a variety of statistical techniques
[a]EC_3
[b]EC_{10}

geometric mean values from field data ranging from 21 to 31 ppm (DW). In contrast, hockey stick analysis of field data on teratogenic effects for stilts showed an egg Se threshold that might be as low as 15.5 ppm. Differences in the threshold estimates for stilts were attributed by Adams et al. (2003) to natural variability in field data. Results of these analyses are similar to the toxicity threshold for egg Se, based on hatchability derived by Ohlendorf (2003). Results of statistical analyses performed by various investigators to assess egg Se concentrations associated with an increased risk of impaired hatchability and embryonic teratogenesis, developed using field and laboratory data for mallard ducks, black-necked stilts, and American avocets, are summarized in Table 9.1.

9.1.3 Selenium Effects on Nesting Avocets and Stilts at the TLDD Evaporation Ponds

To address site-specific concerns at the TLDD, biological and water quality monitoring data from the TLDD Evaporation Basins and the TLDD Compensation Habitat were used to establish statistical relationships between waterborne total recoverable Se and geometric mean egg Se concentrations observed in American

Table 9.2 Predicted waterborne selenium concentration associated with various egg selenium concentrations

	Predicted water Se (mg L^{-1})	
Se in egg (μg g^{-1} dry weight)	American avocet	Black-necked stilt
8	5.5	6.7
10	7.3	8.9
12	9.0	11.3
14	10.7	13.6

avocet and black-necked stilts. Egg Se data from 1994 through 2001 were available for 29 avocet and 44 stilt eggs collected at the North Evaporation Basin, 36 avocet and 33 stilt eggs collected at the South Evaporation Basin, 23 avocet and 8 stilt eggs collected at the Hacienda Evaporation Basin, and 56 avocet and 55 stilt eggs collected at the Compensation Habitat. Waterborne Se concentrations were derived from the annual WDR Water Quality Monitoring Program. Geometric mean egg Se and corresponding waterborne Se concentrations were compared by linear regression and showed a stronger relationship ($r^2 = 0.47$) between the observed geometric mean egg Se for American avocet and waterborne Se at the TLDD facilities than for black-necked stilt eggs ($r^2 = 0.29$). These data are consistent with a review by Adams et al. (2003). Such relatively weak correlations between waterborne Se concentrations and bioaccumulation occur because Se bioaccumulation is primarily a function of uptake of soluble forms of Se (Se (VI) [selenate] and Se (IV) [selenite], in particular). The variation in Se accumulation by avocets and black-neck stilts may also indicate somewhat different food chain transfer pathways. These regression equations were also used to back-calculate the waterborne Se concentration associated with egg Se concentrations ranging from 8 ppm (Ohlendorf and Santolo 1994) to 14 ppm (Adams et al. 2003) (Table 9.2). The regression analyses predicted the site-specific waterborne Se concentrations for the TLDD, based on egg Se thresholds for impaired hatchability from 8 to 14 ppm, to be 5.5 to 13.6 μg Se L^{-1}.

9.1.4 Selenium Effects on Wintering and Migratory Birds

Impacts on birds outside the breeding season associated with dietary Se exposure have not been studied as extensively as have birds during the breeding season, partly because of the difficulty and cost associated with studying these effects (CH2MHill et al. 1993). Laboratory studies on non-breeding mallards have demonstrated both lethal and nonlethal effects, including weight loss, atrophy of feather follicles, and atrophy of lymphoid tissue. It is unknown whether these or other nonlethal impacts on waterbirds during winter contribute to lower reproductive success at breeding grounds. The known effects of Se bioaccumulation in wintering and migratory birds include the following:

Lower overall body weight associated with Se concentration in the liver (Barnum 1992), for green-winged teal, northern shoveler, northern pintail, and ruddy duck. A small (3 %) reduction in body weight associated with bioaccumulation of Se in liver tissue in male and female American coots (Ohlendorf et al. 1988 in studies of the TLDD South Basin and two other sites).

General relationships between Se concentrations >30 mg kg^{-1} in liver and breast and a variety of bird health, survival, and reproduction variables (Beckon and Maurer 2008), with liver Se concentrations <10 mg kg^{-1} reported to be indicative of normal or background Se levels. The report concludes, "hepatic concentrations of selenium are more reliable for delineating populations that are *not* suffering from selenium toxicity than they are for identifying poisoned populations. Elevated levels of hepatic selenium should not be interpreted as anything more than an indication that further study is warranted". It is also likely, based upon variation in life history characteristics and diet, in addition to other factors, that the toxicity thresholds for Se in wintering waterfowl would vary on a species-specific basis, and may also be affected by other biological and environmental conditions. Immunotoxicity of Se has not been studied sufficiently to provide a threshold effect level for waterfowl at the evaporation basins.

Waterbirds not only rapidly accumulate Se, but also rapidly cleanse Se from the body (depurate) when a low-Se diet is available (Heinz et al. 1990). Rapid depuration of accumulated Se would reduce the risk of adverse effects on transient and migratory waterbirds.

The effects of exposure to elevated Se concentrations on wintering and migratory waterfowl within the TLDD South and Hacienda Evaporation Basins are also likely to vary substantially, based upon the duration that the birds are foraging at the evaporation basins and based on the effects of dietary dilution through foraging activity over a more regional landscape. Waterfowl ability to depurate Se suggests that one method of preventing toxic accumulations of Se may be to reduce the duration of exposure to evaporation ponds. Activities such as the intensive hazing program implemented by the TLDD contribute to reducing the potential exposure of these birds to Se at the basins. Other activities at the evaporation basins, such as vegetation control on levees, and the lack of emergent aquatic vegetation, contribute to reduced habitat suitability of the evaporation basin areas for wintering and migratory waterfowl.

Waterfowl hunting in areas adjacent to the South and Hacienda Evaporation Basins during the winter, however, may promote the movement of wintering waterbirds onto the evaporation basins. The Tulare Area Waterfowl Association has entered into an agreement with the TLDD to allow waterfowl hunting on the South Evaporation Basin, which occurs from both installed duck blinds located along the interior basin levees and by boat. The waterfowl hunt program at the South Evaporation Basin serves as an augmentation to the hazing program. One of the goals of the hunt program is to disperse waterfowl from the evaporation basin into adjacent managed wetland areas where waterfowl hunting also occurs.

The Waterfowl Hunt Agreement allows for potential future expansion of the hunt program to include the Hacienda Evaporation Basin, in addition to adjacent floodwater storage areas. The TLDD is continuing to evaluate alternative hazing methods that would contribute to reduced abundance in foraging activity and reduced duration of wintering and migratory waterbird occurrence at the basins to augment or refine the existing hazing program.

In addition to contributing to the effectiveness of the TLDD hazing program, the waterfowl hunt program at the South Evaporation Basin provides local recreational benefits. Recreational waterfowl hunting at the evaporation basin supplements hunting opportunities in the region at existing managed refuges, hunting clubs, and other wetlands in the area.

Based upon results of scientific studies and observations at the TLDD evaporation basins, it was concluded that potential impacts to wintering waterbirds as a result of Se exposure were less than significant. The effects of Se exposure on body weight, condition, and other sublethal indices of health have not been identified to be of a sufficient magnitude to adversely affect the population dynamics of wintering waterbirds. Based on the best available information, the magnitude of sublethal impacts occurring as a result of operation of the TLDD South and Hacienda Evaporation Basins would not be expected to result in a reduction in the populations of wintering waterbirds below self-sustaining levels. Impacts as the result of Se exposure for wintering and migratory waterbirds have been concluded to be less than significant.

9.1.5 Salt Ingestion and Encrustation Effects

As a result of evaporation, salt concentrations increase within individual cells of the evaporation ponds, reaching highest concentrations, which may be hypersaline (e.g., having salt concentrations greater than seawater), in the terminal evaporation basin cells. Concern has been expressed regarding the potential for adverse effects on waterbirds, resulting from salt ingestion or salt encrustation (CH2MHill et al. 1993; Euliss et al. 1989; Barnum 1992; Gordus et al. 2002). Exposure of waterbirds to high salinity has been documented to have adverse effects on them (CH2MHill et al. 1993; Gordus et al. 2002). Effects include the following:

- Ingestion of highly saline water can cause elevated sodium levels within the brain, reduced growth rates, and higher mortality of ducklings (Mitcham and Wobeser 1988; Swanson et al. 1984).
- High sodium concentrations in dead waterbirds within evaporation basins suggest mortality effects (Gordus et al. 2002).

Salt/carbonate encrustation of feathers has been found to inhibit swimming and flying. Encrustation rates in evaporation ponds have been found to range from 5 to 18 % (Euliss et al. 1989) and such encrustation may be associated with salt crystallization on feathers during cold winter months (Euliss et al. 1989; Barnum

1992; Gordus et al. 2002; Wobeser and Howard 1987). The potential indirect effects of salt encrustation on wintering birds and the resulting effects associated with reduced body weight and enlarged salt glands on their subsequent ability to migrate and/or successfully reproduce has been hypothesized, but not investigated. In this context, salt encrustation on wintering waterbirds may act as a sublethal agent adversely affecting subsequent reproductive performance, which, in the absence of any scientific data or investigations, cannot be evaluated with regard to its potential magnitude and significance. Euliss et al. (1989) found the presence or absence of carbonate deposits on ruddy ducks was significantly affected by the month of collection and the pond system, indicating the risk of adverse effects varies on a seasonal basis and within an evaporation basin complex, having cells characterized by a relatively wide range of salinity concentrations, from brackish to hypersaline. Salt encrustation affected tail feathers, causing erosion of the vane, and sometimes only the rachis of the feather remained. Ruddy ducks are less mobile during winter than dabbling ducks (Barnum 1994) and are one of the few species that remain on the TLDD evaporation ponds in high numbers during winter months. Barnum (1992) also observed salt encrustation on other aquatic birds using evaporation ponds, including American avocets and green-winged teal, but did not identify the specific location where waterbirds were collected. Hence, correlations between salinity conditions associated with salt encrustation could not be derived from this study. However, impacts of salt encrustation appear to have greater risks for less mobile, sedentary species, including ruddy ducks and possibly other diving waterbirds that overwinter at evaporation ponds (Gordus et al. 2002).

Lethal and sublethal effects on breeding birds from salt ingestion at the TLDD evaporation basins have not been documented, but the potential for impacts exists, given the high levels of salinity recorded in specific basin cells at the South and Hacienda Evaporation Basins, compared with known levels affecting birds. Water salinity (EC) levels in the terminal cells of the South and Hacienda Evaporation Basins have been greater than levels identified by CH2MHill et al. (1993) to cause lethal and sublethal effects on ducklings. As a result of water management practices, salinities vary substantially among individual cells within each of the three TLDD evaporation basins. Therefore, ducklings and other waterbirds could be exposed to a wide range of salinity conditions from brackish to hypersaline water, depending upon their exposure to an individual cell. Because of the close proximity among evaporation basin cells, waterbirds have the opportunity to readily move from one pond cell to another, thereby having the potential to avoid adverse salinity conditions and/or dilute the effects of adverse salinity by preferentially moving to cells having lower salt concentrations. Water conveyance and supply canals also exist within the immediate vicinity of each of the three evaporation basins, providing additional opportunities for waterbirds to utilize lower saline waters.

Observations made during wildlife abundance and nest surveys have shown that waterfowl may move from higher salinity pond cells to lower salinity areas within an evaporation basin in response to hazing activity and/or habitat preference. Although extensive wildlife monitoring has occurred at the TLDD evaporation basins since 1993, there has been no evidence of salt encrustation on breeding waterbirds during the spring, summer, or fall. However, as discussed below, salt encrustation during the winter has been identified as a factor impacting waterbird survival, requiring additional mitigation measures during winter months to reduce and avoid adverse impacts.

During the spring and summer nesting period (February–August), biological monitoring has not documented lethal or sublethal effects on breeding birds from salt encrustation at the TLDD South and Hacienda Evaporation Basins. These results are consistent with other studies (Euliss et al. 1989; Wobeser and Howard 1987; Gordus et al. 2002) that the risk of salt encrustation varies seasonally in response to air and water temperature. In the absence of documented observations of salt encrustation impacts to breeding birds at the TLDD evaporation basins, or from surveys at other locations (e.g., other SJV evaporation basins subject to intensive biological monitoring programs, Salton Sea, Great Salt Lake, commercial salt production ponds, etc.), the risk of salt encrustation on breeding waterbirds during spring, summer, and fall months is probably minimal.

At a threshold salinity concentration of 70 dS m^{-1}, hypersaline waters may experience salt crystallization as temperatures approach freezing, and the associated risk of salt encrustation on wintering waterbird feathers becomes highest under these conditions (Gordus et al. 2002) for triggering mitigation actions at the Lost Hills Water District evaporation basin.

These general findings regarding waterbird exposure to hypersaline water and the impacts of evaporation ponds can be evaluated, using electrical conductivity (EC) data from each of the individual evaporation pond cells at the North, South, and Hacienda Evaporation Basins. Results of this analysis showed that no evaporation pond cells at the North Evaporation Basin would be expected to result in salt encrustation, since EC levels were consistently below the 8 dS threshold. At the South Evaporation Basin, in two of the terminal cells, ECs exceeded the threshold in which salt encrustation may be a potential risk. At the Hacienda Evaporation Basin, ECs exceeded the 70 dS m^{-1} threshold in three cells. For the five hypersaline ponds, data from the TLDD water quality and biological monitoring program were reviewed and analyzed to assess the potential risk of adverse impacts as a result of salt encrustation. For purposes of this analysis, ruddy ducks were selected as the primary species likely to be affected by salt encrustation, based upon results of earlier investigations (Gordus et al. 2002). Ruddy ducks also are known to occur at the evaporation basins during the wintering period in relatively large numbers and they are relatively sedentary, and hence would be more vulnerable to salt encrustation at the evaporation basins when compared to other species of waterfowl, avocets or stilts. The selection of ruddy ducks for use in the impact assessment of salt encrustation on wintering waterfowl is consistent with the recommendation of Barnum (1992), which concluded, "Among waterfowl wintering on evaporation ponds, ruddy ducks would appear to be the most appropriate on which to concentrate any future research efforts directed towards an assessment of wintering impacts."

Waterbird surveys have been conducted twice per month and provide information on the abundance of each species of bird occurring on the TLDD evaporation basins. Results of these surveys showed that abundance of ruddy ducks on South and Hacienda Evaporation Basin cells varied substantially among surveys and cells. The survey data confirmed, however, that ruddy ducks do occur on the hypersaline cells during winter months and therefore are at risk of adverse impacts resulting from salt encrustation. Monitoring has confirmed that salt encrustation occurs at the hypersaline basins only, and only episodically, with no encrustation observed in 2000 or 2001 and no dead or dying ruddy ducks have been observed during monitoring at either hypersaline or other evaporation ponds.

Salt encrustation impacts at the low-salinity TLDD North Evaporation Basin were found to be less than significant, but impacts resulting from salt encrustation at the South and Hacienda Evaporation Basins were considered to be potentially significant, primarily at the five hypersaline cells. Based upon results of biological monitoring performed at the basins, the magnitude of loss to wintering species, such as ruddy ducks, is expected to be low (<15 birds per day) and to occur on an infrequent basis (episodic events). The risk of salt encrustation affects only a small portion of the ruddy ducks and other wintering waterbirds on the evaporation basins and within the SJV regional area. Although potential adverse effects on individual birds are predicted as a result of salt encrustation, the magnitude of these impacts would not be expected to result in populations of these species being reduced below self-sustaining levels.

9.2 Management Actions to Avoid, Minimize, and Mitigate for Evaporation Basin Impacts to Nesting and Migratory Birds

9.2.1 An Integrated Strategy

Data on both Se bioaccumulation and salt encrustation of waterbirds were the basis for an integrated program to avoid, minimize, and mitigate for potential impacts. The program consists of the following features:

- Modification of evaporation ponds to reduce their suitability for waterbird use;
- Year-round hazing of birds to discourage use of evaporation ponds;
- Measures to remediate and remove Se from the evaporation ponds;
- Measures to break the food chain that results in bioaccumulation of soluble Se;
- Development of alternative, protected nesting bird habitats to divert birds from evaporation ponds; and
- Development of an alternative, low-salinity winter waterfowl habitat.

In simple terms, the integrated strategy is to reduce waterbird nesting and foraging at the South and Hacienda Evaporation Basins, while providing suitable nesting and foraging habitat elsewhere.

Table 9.3 Chronology of TLDD constructed wetland development and modifications to South and Hacienda Evaporation Basins (Davis et al. 2008)

	Activity		
Year	Constructed wetland	South evaporation basin	Hacienda evaporation basin
1994	Construction initiated	Monitoring initiated, tire removal	Monitoring initiated, tire removal
1995	Initial filing, Monitoring initiated	Salt cell removed, windbreak island removal	Na
		Levees reconfigured	Windbreak island removal
1996	On-going management and monitoring	On-going hazing, vegetation control, and disease control	Levees reconfigured and vegetation removed
			On-going hazing, vegetation control, and disease control
1997–2004	On-going management and monitoring	On-going hazing, vegetation control, and disease control	On-going hazing, vegetation control, and disease control

9.2.2 *Evaporation Basin Physical Modifications*

Neutralizing the attractiveness of the evaporations basins as habitat for both waterfowl and shorebirds was optimized for waterbirds, specifically American avocets and black-necked stilts, which nest in open areas with available shallow water for foraging (Table 9.3).

- Increase interior levee slopes and remove tires for levee stabilization;
- Remove windbreak islands and other shallow-water areas;
- Develop inflow control and water depth management to increase water depths within the evaporation basin cells; and
- Control and maintain levee vegetation.

9.2.2.1 Levee Modifications

American avocets, black-necked stilts, and many other waterbirds preferentially forage in shallow-water areas along the levee margins of the evaporation basins. To reduce foraging activity, interior levee slopes were steepened to reduce the availability of shallow-water foraging areas (Table 9.3). In addition, tires used to maintain levee stability were removed to prevent their use as nesting habitat. The steeper and more exposed slopes have experienced substantial wave-induced erosion; thus, some levees require intensive, ongoing mechanical grading to retain the 3:1 levee slope criteria. The change in levee slope maintenance at the evaporation basins has resulted in a significant increase in facility operating and maintenance costs. Although the TLDD has implemented a routine levee maintenance program at both the South and Hacienda Evaporation Basins, fluctuations in water level within the individual pond cells, in addition to localized levee erosion, have contributed to the occurrence of localized areas within some pond cells where shallow-water foraging areas may attract waterbirds. The availability of these potentially suitable foraging areas, however, has been substantially reduced when compared to conditions prior to the TLDD's implementation of the levee modifications.

Although wildlife monitoring is conducted at the evaporation basins and provides information on the overall performance of the program in reducing and avoiding adverse impacts to nesting waterbirds, the monitoring program does not provide detailed information for use in evaluating the incremental effectiveness of the individual actions, such as levee modifications, in modifying bird usage patterns within the evaporation basins. As part of the overall program, TLDD is continuing to investigate and evaluate methods for improving the effectiveness of various actions implemented at the evaporation basins.

9.2.2.2 Windbreak Removal

Results of previous biological surveys had shown large numbers of waterbirds nesting and foraging on shallow, windbreak islands within the South and Hacienda Evaporation Basins. Nests on the windbreak islands also had substantially lower risk of mortality from predators. During the initial construction of the evaporation basins, shallow island areas were created to act as windbreaks and reduce wave-induced erosion of interior levees. In an effort to reduce waterbird foraging and nesting within the evaporation basins on these shallow windbreak islands, all windbreaks within the South and Hacienda Evaporation Basins were removed.

9.2.2.3 Inflow Distribution and Control Structures to Manage Depth

Maintaining water depths of 0.61 m (2 ft) or greater limits avocets and stilts access to forge. This is being accomplished with modifications to the inflow distribution and control structures at the South and Hacienda Evaporation Basins. The inter-cell delivery system and the management of drainage water within the cells allows maintenance of a depth of 0.61 m (2 ft) (except when draining or filling). Maintenance of water depths $>$2 ft within pond cells has been incorporated as part of routine evaporation basin operations. Data from hazing (see below) and wildlife monitoring are used on a routine basis to identify the distribution of waterbirds within the evaporation basins and to identify specific locations where additional physical modifications and/or focused hazing activity may be required to reduce further waterbird foraging and nesting activity.

9.2.2.4 Vegetation Control

The TLDD conducts an annual program of vegetation removal using mechanical sloping, pre-emergent and contact herbicides, coupled with regular levee maintenance to minimize plant growth (especially perennial plants such as *Allenrolfea spp.*), which may offer nesting cover for ducks, blackbirds, and shorebirds.

Vegetation control is part of routine TLDD maintenance at the evaporation basins and is scheduled regularly to avoid potential adverse effects to nesting waterbirds. To reduce impacts to nesting birds, levee vegetation control and maintenance activities are scheduled, to the extent possible, to avoid disruption during the nesting season.

9.2.3 Hazing

Hazing, creating a disturbance regime to make the evaporation ponds less attractive to birds, is managed both directly and indirectly. Hazing methods have evolved, based on results of annual biological monitoring. The direct, year-round hazing program, developed jointly by TLDD operators and wildlife biologists responsible for waterbird abundance and nest monitoring, includes the use of Hovercraft to facilitate hazing within both the South and Hacienda Evaporation Basins to complement hazing from perimeter and interior levees using all-terrain vehicles (ATV) and cracker shells. In addition to capital costs, TLDD has hired additional employees to perform the enhanced hazing program. As an augmentation, TLDD implemented a program using foil reflector tape on stakes placed at approximately 3.05–4.57 m (10–15 ft) intervals in areas of observed pre-nesting and nesting attempt activities at both the South and Hacienda Evaporation Basins. The TLDD also uses portable propane cannons to augment the basic hazing program. In addition, the close proximity of hypersaline pond cells to cells and canals within the basin complex having substantially lower-salt water provides migratory birds removed from the hypersaline cells with alternative foraging areas. The availability of undisturbed areas with low-salt water enhances the net effect of hazing. Movement of birds, in response to intensive hazing, from hypersaline waters to adjacent cells having lower salinity would reduce or eliminate salt accumulations on feathers.

The hazing program has been designed to reduce and avoid nesting activity at the South and Hacienda Evaporation Basins, while also recognizing the importance of avoiding disturbance and potential adverse impacts to nesting waterbirds in the event that nests are established within hazing areas. If waterbird nests are established at a specific location within an evaporation basin pond cell, hazing is discontinued in the vicinity of the established nest to avoid disruption and impacts to developing chicks. Once nesting activity is complete (e.g., a nest has been abandoned or destroyed by predators or chicks have successfully fledged, etc.), the nest is no longer considered to be active and hazing in the area is resumed.

Indirect hazing also occurs in conjunction with other management activities. For example, Novalek, Inc. commercially harvests brine shrimp from the TLDD South and Hacienda Evaporation Basins. Brine shrimp harvest is performed year-round, with harvest activities concentrated in the hypersaline cells within the basins where brine shrimp production is greatest. Commercial brine shrimp harvest is performed from a barge equipped with an outboard motor, which circulates within the hypersaline cells, typically during the late night and early morning hours, while filtering and processing brine shrimp. Brine shrimp harvest activities, which have not occurred historically within the TLDD evaporation basins, augment the TLDD hazing program, with specific emphasis on those evaporation pond cells where the risk of salt encrustation to wintering waterbirds is greatest.

Indirect hazing also occurs as a result of waterfowl hunting. The California Department of Fish and Wildlife (CDFW) collected breast muscle tissue samples from waterfowl at the South and Hacienda Evaporation Basins during the fall of 1997 and 2000 for use in determining Se concentrations. Breast tissue Se concentrations observed during both years were considered by CDFW to represent a low risk of adverse effects to human health associated with waterfowl hunting at the South and Hacienda Evaporation Basins. Based on these results, the CDFW approved a waterfowl hunt program on the TLDD South Evaporation Basin and in 2001, TLDD entered into an agreement with the Tulare Basin Wetlands Association to allow hunting on the South Evaporation Basin (the Waterfowl Hunt Agreement) beginning in Fall 2001. The Waterfowl Hunt Agreement allows for future expansion of the hunt program to the Hacienda Evaporation Basin, Hacienda floodwater storage area, and South Wilbur floodwater storage area.

9.2.4 Breaking the Bioaccumulation Food Chain

The evaporation basins, particularly the terminal basins with high salinity, are high nutrient environments that support the development of invertebrates. The most common invertebrates are backswimmers (*Notonecta unifasciata*), water boatman (family *Corixidae*), brine flies (family *Ephydridae*), damselflies (order *Odonata*), and brine shrimp, family *Artemiidae* (Hanson 1993). Brine shrimp make up a large component of the invertebrate biomass and represent a concentration pathway for bioaccumulation. Since 1998, TLDD has managed this potential problem by harvesting brine shrimp and selling them, thus effectively breaking the food chain and removing Se from the pond environment. In addition, preliminary research on Se cycling within the TLDD evaporation basins suggests that commercial brine shrimp harvest may result in changes in algal populations and dynamics within the cell that result in an increase in Se volatilization, which further contributes to a reduction in waterborne Se within the cell.

Annual average brine shrimp harvest typically ranges from 90,718 to 136,008 kg (200,000–300,000 lb) of commercial grade fish food per year. The highest levels of harvest occur during the spring and summer months (March through October). The brine shrimp harvested by an independent contractor from TLDD evaporation basins are sold commercially worldwide. Tests using tropical aquarium fish fed brine shrimp from the TLDD evaporation basins have shown no adverse effects on adult survival or reproductive success. Opportunities to increase brine shrimp production through the use of nutrient enhancement and hyper-saline water source control are being explored.

9.3 Measures to Avoid and Minimize Salt Encrustation of Wintering Birds at the South and Hacienda Evaporation Basins

To mitigate the risk of salt encrustation in wintering waterbirds, the TLDD implemented a plan designed specifically to minimize and avoid potential impacts to them as a result of salt encrustation and sodium toxicity. The avoidance program, including brine shrimp harvest, occurs within the five hypersaline pond cells at the South and Hacienda Evaporation Basins, triggered when air temperatures (typically, at night) decline rapidly to 2.2 °C (36 °F) or below, based on CIMIS Station observations. Information available daily on predicted chill days (as part of agricultural weather reports) is also used in alerting and triggering avoidance actions. Observations of waterbird activity on hypersaline cells by TLDD staff, during biological monitoring, and observations and information provided by brine shrimp harvest crews is also used as input to decisions about when to initiate the avoidance actions.

On those winter days (December–February) when air temperatures decline to 2.2 °C (36 °F), TLDD mobilizes staff to search the hypersaline cells in the South and Hacienda Evaporation Basins for birds affected by salt encrustation. The searches are done from both the levees and by boat. Searching the identified hypersaline evaporation basin cells by boat and from shore provides good coverage of the potentially affected area and facilitates the ability to detect and capture waterbirds affected by salt encrustation that may be present at the facility. If field observations detect birds affected by salt encrustation, a TLDD crew or contractor immediately begins capturing affected waterbirds using large hand-held nets. The captured waterbirds are immediately relocated and released into low saline ponds or other waters within the evaporation basin. Capture and release of waterbirds are conducted to reduce handling stress, to the extent possible. Waterbirds are released into low-salinity areas where encrusted salts dissolve and return to solution. Only under extreme conditions would a captured bird be held and the feathers rinsed or washed to remove salt deposits. Birds are captured and released in accordance with terms and conditions of the CDFW Scientific Collection Permit. The mitigation actions implemented at the TLDD evaporation basins are consistent with the actions and triggering conditions identified for other similar mitigation programs designed to reduce and avoid potential impacts of salt encrustation at evaporation basins (Gordus et al. 2002).

9.4 Compensation Wetland Habitat for Unavoidable Shorebird Losses During the Spring-Summer Foraging and Nesting Period

Although the avoidance and minimization measures implemented as part of this program have proven to be extremely effective, unavoidable losses of wildlife are predicted to occur as a result of factors, such as exposure to elevated Se concentrations, physical hazards associated with maintenance activity, and nest flooding resulting from water level fluctuations within the evaporation pond cells. These unavoidable impacts require further mitigation in the form of an off-site wetland designed and managed to compensate for residual impacts to shorebird foraging and nesting during the spring and early summer.

Hanson (1993) prepared a protocol for assessing the compensation habitat requirements to mitigate for the expected level of unavoidable losses resulting from operation of the TLDD South and Hacienda Evaporation Basins. The protocol used the available data to estimate the potential effectiveness of both the avoidance and mitigation measures proposed for implementation at the evaporation basins and the predicted effectiveness and use of a constructed and seasonally managed wetland habitat for nesting by stilts and avocets. Based on the 1993 protocol, Hanson (1993) estimated that a compensation wetland habitat of 84 ha (207 acres) would be required to fully mitigate unavoidable TLDD evaporation basin losses. Based on this original estimate, the TLDD constructed a 167-ha (288-acre) managed wetland specifically designed and operated to provide foraging and nesting habitat for American avocets and black-necked stilts, but also benefiting other wildlife. Subsequently, the USFWS (1995) developed two alternative protocols for use in estimating compensation habitat requirements (the USFWS Hen-Wise protocol and USFWS Egg-Wise protocol) that were based on the proportion of waterbird eggs having various Se concentrations. Hanson (1995) refined a protocol that used results of more recent biological surveys and integrated concepts included in the USFWS (1995) protocols to develop revised estimates of the wetland habitat required to mitigate fully unavoidable impacts to stilts and avocets. The analytic frameworks provided an initial basis for estimating compensation habitat requirements to mitigate for evaporation basin operations. In the absence of actual post-implementation monitoring data, the protocols relied on conservative assumptions regarding the performance and effectiveness of actions implemented at the evaporation basins and at a constructed and managed wetland. Long-term results from more than 10 years of monitoring at the evaporation basins and compensation wetland habitat have provided new insights into the performance of these measures and have been used to revisit the earlier protocols with more reliable assumptions and relationships to recalculate the estimates of compensation wetland habitat requirements.

9.4.1 Design, Management, and Performance of Seasonal Wetlands for Evaporation Basin Mitigation

9.4.1.1 Habitat Design

The 117-ha (288-acre) compensation wetland habitat was designed to provide foraging and nesting habitat for black-necked stilts and American avocets and thus involved creating a habitat with shallow, open water for foraging, a vegetated wetland edge for stilt nesting, and bare upland areas for avocet nesting. American avocet and black-necked stilts inhabit saline areas where production of macroinvertebrates is high and they prefer a gradually sloped shoreline to a more abrupt shoreline. The design for the constructed wetland incorporated a 2:1 ratio of foraging habitat to nesting habitat and water depths of 10–16 cm (4–6 in.). The constructed wetland includes low-profile lanes (shoreline) immediately adjacent to extensive shallow water areas to support macroinvertebrate production and provide extensive foraging habitat. There were 34 lanes, each approximately 1.6 km (1 mile) long, providing over 59.5 km (37 miles) of nesting habitat approximately 9.75 m (32 ft) wide and wetted foraging areas approximately 19.5 m (64 ft) wide, with gentle shoreline slope (10–12H:1V). The wetland provides for continuous water flow; the depth of the water is self-regulating; and there is no terminal ponding. This constant flow of water through the wetland is intended to reduce impacts of evaporation on water quality and avian disease. To attract and maintain a population of nesting shorebirds, invertebrate breeding ponds were created throughout the interior of the wetland habitat. These invertebrate ponds were located at the bottom of the foraging areas approximately 45.7 cm (18 in.) deeper than the surrounding channel areas. Each invertebrate pond was approximately 2.1 m (7 ft) wide and 21.7 m (50 ft) long, spaced at approximately 0.16 km (0.1-mile) intervals, with a total of 8–10 invertebrate ponds per lane. The wetland was completely encircled by an electrified perimeter fence (Mayer and Ryan 1991), primarily to exclude coyotes (*Canis latrans*) and other terrestrial predators.

9.4.1.2 Operation

The wetland is typically flooded in late February, with water supplies maintained through the completion of the nesting season, which typically ends in early August. The wetland is then drained, and it remains dry from September through February. While dry, maintenance activity is performed, including vegetation control by discing, and pre-emergent herbicide application along the nesting lanes. To avoid potential effects of herbicides on nesting success, pre-emergent herbicide application is made when the habitat is dry using localized spot application in accordance with manufacturer directions, and allowing sufficient time between applications and nesting for the active ingredients to denature to a non-toxic level. The water supplied to the ponds is saline agricultural drainage water having low Se concentrations (typically <2 μg Se L^{-1}), freshwater supplies, or a blend of freshwater and saline waters. The selection of a water supply was based, in part, upon the seasonal availability of freshwater and on Se concentrations in the drainage water sources. Other water supply considerations included the following:

- Many of the waterbird species of interest inhabit coastal marine areas and are adapted to saline waters.
- Macroinvertebrates, which provide the forage base for many of these species, also occur in relatively high abundance in saline waters.

Saline agricultural drainage water, provided it had low Se concentrations, could thus be used beneficially as a water supply source for the wetland habitat. Saline drainwater was approved under the 1993 WDR and has been used as the sole water supply source for the TLDD Compensation Wetland Habitat since 1999. TLDD routinely monitors Se concentrations in the water supply to the Compensation Habitat during the spring and summer operation period. Detailed monitoring of EC and Se has been used to manage saline drainage water and to blend saline drainage water with freshwater within the Compensation Wetland Habitat. The use of saline drainage water within the TLDD Compensation Wetland Habitat, on the condition that it satisfies specific water quality criteria designed to avoid adverse impacts on wildlife, is consistent with California water policy and the state Water Code that reasonable, beneficial use of water be maximized.

9.4.2 Wetland Habitat Performance

The TLDD Compensation Wetland Habitat was operated using only low-Se saline agricultural drainage water supplies during 1999–2004. Electrical conductivity at the inlet to the wetland ranged from approximately 5–10 dS m^{-1}. During passage through the wetland, EC increased as a result of evaporation to approximately 15–20 dS m^{-1} in water discharged from the wetland.

The geometric mean Se concentration of the agricultural drainage water used to support the wetland was 1.4 μg Se L^{-1} (Table 9.4). Annual geometric mean Se concentrations ranged from 0.9 to 2.0 μg Se L^{-1}. We observed a maximum Se concentration of 23 μg Se L^{-1} on one sampling occasion, but this atypical result may reflect sampling or analytical artifacts because the Se concentration we measured in the subsequent sampling was 2.0 μg Se L^{-1}. In contrast to salinity, Se did not show a pattern of increasing concentrations between the inflow and discharge from the wetland.

Annual geometric mean Se concentrations in American avocet eggs (Table 9.5) ranged from 2.0 to 5.3 μg Se g^{-1} dry weight, with an overall geometric mean of 2.8 μg Se g^{-1} (sample number, $n = 70$). Annual geometric mean Se concentrations for black-necked stilt eggs (Table 9.5) ranged from 2.2 to 5.1 μg Se g^{-1}, with an overall geometric mean of 2.9 μg Se g^{-1} ($n = 67$). Egg Se concentrations for American avocets and black-necked stilts were not significantly different ($P > 0.05$). Maximum egg Se concentrations at the wetland were below thresholds for adverse effects and substantially below the values reported for the evaporation basins (Table 9.5). Eggs collected from the constructed wetland showed no embryonic abnormalities for either American avocet ($n = 70$) or black-necked stilt ($n = 67$).

The constructed wetland was used for nesting and foraging by a variety of species including avocets and stilts, snowy plover, killdeer, various ducks, and upland species (Table 9.6). Nesting by 13 taxa has been documented at the wetland, although, on average, nesting activity by American avocet and black-necked stilt accounted for 91 % of the nesting activity at the site (Table 9.6; Fig. 9.1). Over the 10-year monitoring period, average nest density for American avocet and black-necked stilts was approximately 24.8 nest attempts per ha (10.1 nests/acre; overall annual $\bar{x}$ of 2,896 nest attempts within the 117-ha (288-acre) wetland). The highest nesting density observed for avocets and stilts occurred in 1998

Table 9.4 Waterborne total selenium concentrations (μg L^{-1}) in the TLDD compensation wetland (Davis et al. 2008)

			Selenium concentration (μg L^{-1})		
Year	Sampling period	Number of sample	Geometric mean	Minimum	Maximum
1995	29 March–31 July	11	1.1	<0.5	23.0[a]
1996	22 March–10 July	24	0.9	<0.5	3.0
1997	5 March–15 July	69	1.4	<0.5	3.3
1998	1 April–1 June	21	2.0	1.2	4.2
1999	6 April–1 June	21	1.2	0.7	2.7
2000	1 April–1 June	14	1.4	1.2	1.8
2001	2 April–5 June	21	1.9	<0.5	3.2
2002	9 April–12 June	21	2.0	1.3	3.0
2003	30 April	3	1.2	0.8	1.9
2004	7 April–9 June	4	1.6	1.4	2.2
1995–2004	5 March–31 July	209	1.4	0.7	4.2

[a]Based on other data, this is a probable sampling error

Table 9.5 Selenium concentrations (μg g^{-1} dry weight) in American avocet and black-necked stilt eggs collected at the TLDD compensation wetland, 1995–2004 (Davis et al. 2008)

	Se in American avocet egg (μg g^{-1} dry wt.)				Se in black-necked stilt (μg g^{-1} dry wt.)			
Year	n	Geometric mean	Minimum	Maximum	n	Geometric mean	Minimum	Maximum
1995	8	5.3	4.3	6.2	8	4.1	3.4	5.5
1996	8	2.7	2.2	3.1	8	3.1	2.2	4.1
1997	8	3.0	2.2	4.0	8	3.5	1.9	5.6
1998	8	2.8	1.7	9.3	7	2.4	1.2	3.0
1999	8	2.0	0.6	3.7	8	5.1	3.7	9.7
2000	8	2.0	1.3	3.1	8	2.4	1.5	3.7
2001	8	2.7	1.3	5.7	8	2.2	1.7	3.2
2002	8	2.2	1.2	4.6	8	2.3	1.7	2.7
2003	3	4.5	2.9	6.9	2	3.0	2.5	3.8
2004	3	2.7	2.3	3.3	2	2.5	2.0	3.0
1995–2004	70	2.8	0.6	9.3	67	2.9	1.2	9.7

n denotes number of samples

(3,834 nests; 32.8 nests/ha and 13.3 nests/acre) and the lowest number of nesting attempts occurred in 1996 (1,576 nests; 13.5 nests/ha and 5.5 nests/acre).

American avocets and black-necked stilts inhabited the compensation wetland beginning in March, with nesting extending through July. Abundance of both avocet and stilts increased during the late spring when birds were actively foraging, mating, and nesting within the wetland. The peak in abundance, which typically occurred from approximately mid-May through June, reflects the occurrence of both nesting adults and the recruitment of fledglings to the population inhabiting the area.

Results of nest-fate monitoring (Table 9.7) have shown high nest success, with an overall average of 82 % of the American avocet nests and 75 % of the black-necked stilt nests classified as hatched or presumed hatched. Abandoned nests averaged 10 % for avocet and 16 % for stilts. Nest losses attributable to predation

Table 9.6 Estimated numbers of nests for 14 avian species that utilized the TLDD compensation wetland, location, 1995–2004 (Davis et al. 2008)

	Estimated number of nests[a]									
Taxa	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004
American avocet	754	829	1,631	2,052	1,362	1,374	2,442	1,631	1,362	2,387
Black-necked stilt	1,303	747	1,539	1,782	1,272	996	1,348	1,262	1,665	1,220
Snowy plover	41	41	77	120	60	48	48	36		
Killdeer	42	68	145	216	258	144	180	144	180	324
Mallard		2	4	18	6	6		12		
Northern pintail	2	6	2	30	6	18	24	12	72	36
Cinnamon teal		1	5	6		12	6	24		
Gadwall		1	1				6			
Unidentified duck	2	12	5	24	18	12	24	48	72	108
Horned lark	4	7	6							
Morning dove	0	1								
Wilson's phalarope							12			
Forster's tern							6			
Northern shoveler							6	12	36	
Total	2,560	1,932	3,735	4,522	3,157	2,804	4,102	3,181	3,387	4,075

[a]We adjusted nest start estimates for sampling effort. We adjusted American avocet and black-necked stilt nest starts using a Mayfield correction

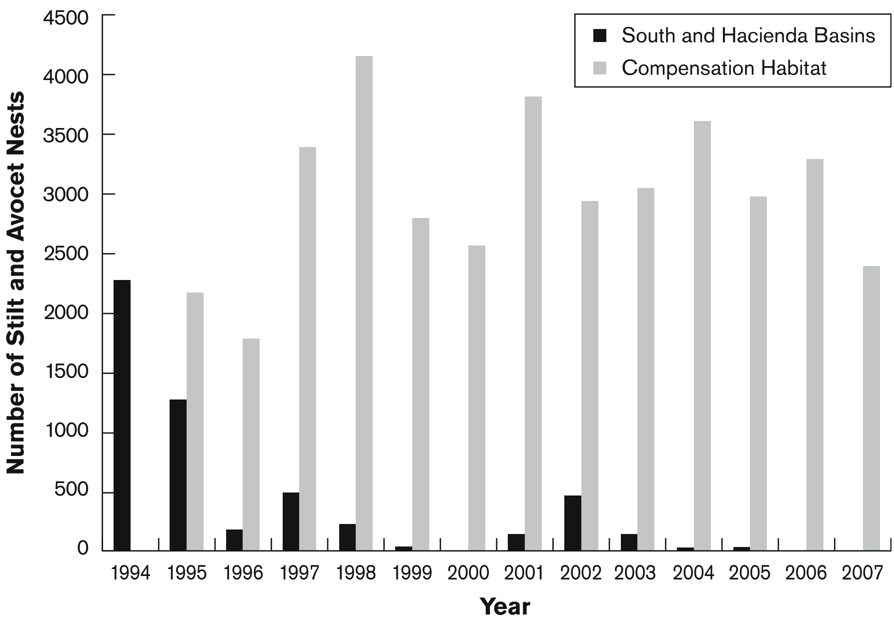

Fig. 9.1 American avocet and black-necked stilt nest starts at the drainage water evaporation basins (South and Hacienda) verse the compensation habitat, 1994 to 2007

Table 9.7 Fate of nests for American avocet and black-necked stilt at the TLDD compensation wetland, 1995–2004 (Davis et al. 2008)

Year	Hatched or presumed hatched[a]	Abandoned unknown	Lost or unknown	Destroyed predator	Destroyed or flooding unknown	Past term or nonviable
American avocet[b] (%)						
1995	65	14	20	<1	<1	0
1996	75	13	10	2	<1	<1
1997	84	6	7	<1	2	0
1998	86	4	10	0	<1	0
1999	93	6	4	<1	0	0
2000	90	4	4	0	<1	<1
2001	78	16	6	<1	0	0
2002	87	9	4	0	0	0
2003	76	22	2	0	0	0
2004	86	7	8	0	0	0
Mean	82	10	8	<1	<1	<1
Black-necked stilt %[b]						
1995	63	19	17	<1	<1	<1
1996	72	17	8	2	2	0
1997	75	14	16	1	2	<1
1998	84	6	10	<1	0	0
1999	82	13	4	<1	1	0
2000	79	15	5	0	0	1
2001	76	18	6	0	0	1
2002	72	22	6	0	0	0
2003	70	21	9	0	0	0
2004	81	10	10	0	0	0
Mean	75	16	9	<1	<1	<1

[a]At least one egg hatched
[b]Percentages may not sum to 100 % due to rounding

mortality, nest flooding, destroyed nests, and past term nonviable nests were consistently >1 % each year. Hothem (1989) reported results of waterbird nesting within the south Grasslands area. Nest fate surveys were conducted, which included data on both American avocet and black-necked stilts, in addition to killdeer and several species of waterfowl. The percentage of avocet and stilt nests classified as hatched at control sites in 1986 surveys was 31 and 25 %, respectively. Predation accounted for the largest percentage of nest failures for both avocets (66 %) and stilts (64 %) in these surveys. The percentage of avocet and stilt nests classified as hatched or presumed hatched at the TLDD Compensation Habitat was substantially greater (overall nesting success averaged 79 %), when compared to the overall average avocet and stilt nesting success in the Grasslands surveys (29 %). The substantial reduction in nest losses as a result of predation at the TLDD Compensation Habitat appears to be a major factor in the high levels of nesting success by both avocets and stilts. The density of nests at the Compensation Habitat from 1996 through 2004 was approximately 300 times that of the nests at the South and Hacienda Evaporation Basins (Table 9.8).

Table 9.8 Comparison of American avocet and black-necked stilt nest starts at the TLDD South and Hacienda Evaporation Basins and compensation wetland habitat

	Mayfield adjusted American avocet and black-necked stilt nest estimates													
Location	1994	1995	1996	1997	1998	1999	2000[a]	2001[b]	2002[b]	2003[b]	2004[a]	2005[a]	2006[a]	2007[a]
South Evaporation Basin	872	755	110	462	168	15	1	107	437	104	1	7	2	0
Hacienda Evaporation Basin	1,394	497	65	5	50	9	1	1	19	5	8	1	0	1
Total Evaporation Basin	2,266	1,252	175	467	218	24	2	108	456	109	9	8	2	1
Compensation habitat	–	2,165	1,771	3,372	4,123	2,773	2,561	3,790	2,893	3,027	3,607	2,968	3,276	2,376

[a]Numbers of stilts and avocets nesting at the South and Hacienda evaporation basins were insufficient to calculate Mayfield adjustment factors: nest numbers reported are the actual nest numbers observed

[b]Numbers of nesting stilts and avocets at the Hacienda Evaporation Basin were insufficient to calculate Mayfield adjustment factors: nest number reported is the actual number observed

9.5 Compensation Habitat for Impacts to Winter Waterfowl Habitat

In 2005, TLDD identified a 123-ha (305-acre) site northwest of Corcoran, California as a location for a winter wetland habitat site using saline agricultural water supplies. The site had been in active agricultural production for several decades. This site has saline-alkali soils typical of lower SJV floodplains and lakebeds, with high clay content and thus poor drainage and low permeability. There is low-Se saline subsurface drainage water available to this site during the period when winter waterfowl are present in the region for use as a water supply to the wetland. There are wetland vegetation types suitable for the target species that will readily germinate and grow on the soils of the site and in water with a mean salinity from 3,000 mg L^{-1} to 4,000 mg L^{-1} TDS. The wetland was designed to benefit dabbling and diving ducks by providing wintering wetland habitat and islands for loafing.

9.5.1 Habitat Design

The winter waterfowl habitat was designed based on a review of the design and performance of similar habitats at Suisun March, which uses water of similar salinity to that available in the TLDD, and the Kern National Wildlife Refuge (KNWR), which has a similar place in the migratory bird flyway. The 123 ha (305 acre) wetland was designed to provide habitat for species commonly observed at the KNWR, primarily dabbling and diving ducks including green-winged teal (*Anas carolinensis*), mallard (*Anas platyrhynchos*), northern pintail (*Anas acuta*), northern shoveler (*Anas clypeata*), gadwall (*Anas strepera*), and American widgeon (*Anas americana*) and also the less common cinnamon teal (*Anas cyanoptera*), ring-necked duck (*Aythya collaris*), and ruddy duck. Other common winter visitants to Kern National Wildlife Refuge include great blue heron (*Ardea herodias*), black-crowned night heron (*Nycticorax nycticorax*), killdeer (*Charadrius vociferus*), black-necked stilt, ring-billed gull (*Larus delawarensis*) and California gull. The design incorporated some provisions for occasional visits by geese, bitterns, herons, egrets, and cranes. Given that some shorebirds may overwinter in the region, shallow-water habitats for species such as killdeer and black-necked stilts were also provided. The design provided for water depth of about 15–46 cm (6–18 in.) (SRCD 1998; CWA 2005).

The TLDD habitat was designed with three separate hydrologic units facilitating flooding and draining at different times and at different rates to create a mosaic of habitats and reflect the needs of target species. Gentle 10:1 slopes form the perimeter levees around each of the three hydrologic units. Water depths are varied to reflect the needs of different species. Numerous loafing islands were created in each cell to provide a variety of habitats. Each cell can be operated independently to provide a variety of water depths. Water application and dry down can be timed independently to facilitate different types of seed germination and plant growth.

9.5.2 Water Quality

Water quality within the managed wetland is important for several reasons. First, poor water quality may affect the health of target species. Second, water quality may affect the growth of vegetation and/or macroinvertebrates and thus influence food supplies for target species. Third, some species prefer low turbidity water for foraging. Salinity within the wetlands is maintained within a range from 3,500 to 5,000 mg l^{-1} to provide conditions suitable for germination and growth of key seasonal wetland plants: alkali bulrush (*Scirpus robustus*), fat hen (*Atriplex triangularis*), cattails (*Typha* ssp.), tules (*Scirpus acutus*), watergrass (*Echinochloa crusgalli*), smartweed (*Polygonum lapathifolium*), and swamp timothy, *Heleochloa schenoides* (CWA 2005). Salinity effects are greatest on germinating plants; soil leaching to reduce soil salinity prior to planting in the spring is necessary. The optimal germination of key aquatic plants for seasonal wetlands occurs well below the salinity tolerance of the adult plants. The wetland water supplies are managed within a range considered to be suitable for propagation and growth of desirable plant species to provide both cover and forage for wintering waterfowl. Bioaccumulation of some Se and other constituents in subsurface agricultural drainage water would have the greatest effects on waterfowl that feed within the wetland environment, as opposed to birds like geese, which forage on grains in adjacent farm fields. The wetland was designed and is operated as a flow-through facility that reduces the potential for such adverse effects.

Several target species, notably pintail ducks, prefer clear water for foraging and thus it was important to construct a marsh with a settling basin to minimize turbidity and suspended sediment concentrations in the main wetland. The burrow areas adjacent to the levees provide a settling basin to reduce turbidity in the habitat.

9.5.3 Water Management for Plant Production

Seasonal wetlands depend on moist soil management to provide an abundance and diversity of seeds for waterfowl. CWA (2005) reported that commercial grains in adjacent farmlands lack the vitamins, minerals, and proteins required for waterfowl survival and subsequent reproductive success. For a diverse assemblage of waterfowl, including dabbling and diving ducks as well as incidental use by other waterfowl, such as cranes and geese, CWA (2005) suggests a diverse plant community including seed species and species that meet waterfowl needs for cover and thermal protection, as well as enhancing invertebrate production.

Water management is timed to reflect the arrival and departure of the target bird species, which may vary by location and from year-to-year. CWA (2005) recommends early fall flooding, if possible, for locally reared mallards and early migrant pintails. Flooding and maintenance of ponds may involve flooding and drawdown for leaching followed by re-flooding. Ponds generally remain flooded from approximately September through early March (depending on the year, observed conditions, and target species behavior). Pond water is circulated using the flow-through system for salinity control and production of aquatic invertebrates.

Timing of the drawdowns coincides with optimum germination conditions (primarily soil temperature) and periodic discing to maintain the successional stage required for the target vegetation (CWA 2005). The rate of drawdown also affects moist-soil plant composition, seed production, soil-salt levels, and the duration of food availability for waterfowl (CWA 2005). Slow drawdowns concentrate aquatic invertebrates, allow for greater plant community diversity, and if completed in mid-spring, may enhance seed production. Rapid drawdowns enhance leaching and improve soil salinity conditions. Because different forage plants require different cultivation practices, a wetland complex with variable water management regimes for various portions of the overall wetland may be desirable.

The TLDD winter waterfowl habitat is flooded to provide maximum benefit to a broad cross-section of waterfowl. One unit is typically flooded in September, one in October, and one in November to provide early-, mid-, and late-season habitat. The habitat is typically dried down in March.

9.5.4 Invertebrate Production

Miller et al. (2000) document that invertebrate consumption accounts for 25–30 % of total food intake by volume for mallards, gadwalls, and widgeons, with agricultural crops amounting to 45–60 % of total food intake for these three waterfowl. In the absence of adjacent foraging areas, the wetland must sustain the birds, which utilize it. Wetland productivity and function can therefore be enhanced significantly by (a) effective production of invertebrates and (b) grain production in adjacent areas or supplementation on site. During the transition from winter to spring, female ducks may shift from a plant diet to a diet focused on aquatic invertebrates to enhance the protein content of their diet and support egg formation and growth. Wetland design affects invertebrate production, depending on the group of invertebrates involved and their life history strategies (Eldridge 1990).

The saline water supply used at the TLDD wetland habitat is nutrient rich and provides an abundant diet for invertebrates. The deeper areas of the cells provide habitat for invertebrates and enhance the potential for the wetland to provide high protein food sources for female ducks, as they prepare for migration to summer breeding grounds. Because mosquitoes are not desired aquatic invertebrates, the wetland complex has been designed and managed to avoid conditions that would result in abundant, exposed mud in spring and summer.

Table 9.9 Bird use surveys conducted at the 123-ha TLDD saline winter wetland habitat

	Bird survey date		
Species	Dec. 15, 2006	Jan. 19, 2007	Feb. 16, 2007
Grebes	30	0	0
Waders	1	1	0
Geese	0	0	0
Dabbling ducks	160	315	94
Diving ducks	2,000	80	214
Waterfowl	2,160	395	308
Hawks and falcons	0	0	2
Rails and moorhens	0	0	0
Coots	122	1,200	70
Plovers	0	17	3
Recurvirostrids	80	65	10
Sandpipers	1,192	201	860
Shorebirds	1,272	283	873
Non-recurv. shorebirds	1,192	218	863
Gulls	0	0	0
Terns	0	0	0
Total water birds	3,586	1,880	1,254
Total land birds	0	18	26
Grand total	3,586	1,898	1,280

9.5.5 Vegetation Management

Seasonal maintenance is performed during the summer months when the habitat is dry. A stubble disc followed by a finish disc is used to control dense stands of tules and cattails vegetation. Finish disking takes place prior to fall flooding to break up dense stands of moist soil and plants, creating diverse aquatic habitats. Herbicides are avoided to the extent feasible, except along the levees. Mowing, burning, and disking are used periodically to manage very dense vegetation.

9.5.6 Wildlife Use at the Winter Habitat

Biological monitoring was conducted during one day each month at the winter wetland habitat in December 2006, January, 2007, and February 2007. A total of 27 taxa were observed at the wetland with American coot (*Fulica americana*), northern shoveler, western sandpiper, least sandpiper, and long-billed dowitchers being the most abundant. Waterbirds represented 99 % of the birds observed during the three surveys (Table 9.9).

9.5.7 *Effectiveness of the Program*

9.5.7.1 Evaporation Pond Reconfiguration and Hazing

Routine maintenance of the levees, including vegetation control and removal, has coincided with a substantial reduction in nesting by ducks at the South and Hacienda Evaporation Basins. Since modifications to the South and Hacienda Evaporation Basins were completed, which included enhanced vegetation control along levees, only three duck nests have been observed during the past 10 years. During the same time period (1996–2004), a total of 76 duck nests have been observed at the Compensation Habitat, of which 39 nests were classified as hatched or presumed hatched. From 1994 through 2004, the number of breeding waterbirds, such as American avocet and black-necked stilt, at the South and Hacienda Evaporation Basins declined by about 93 % (Table 9.10, Fig. 9.1).

It has been hypothesized that during wet years, such as 1997 and 1998, when adjacent freshwater habitat from flooding is available within the areas adjacent to the South and Hacienda Evaporation Basins, that the abundance of nesting waterbirds would be decreased at the basins and that this may account for some of the observed declines in waterbird use of the evaporation basins. In contrast, during dry years, when adjacent freshwater habitat is not available, it has been hypothesized that waterbird nesting at the evaporation basins would be expected to increase. During the flood years of 1997 and 1998 resulting in a net 24,807 ha (61,300 acres) of floodwater storage on May 1, 1997, and 17,037 ha (41,200 acres) of net flood storage on May 3, 1998, nest monitoring at the evaporation basins counted the highest nest abundance by avocets and stilts (467 nests during 1997, 218 nests in 1998) during the 1996–2006 period. During the dry years of 1999 and 2000, when no adjacent freshwater habitat was created by floodwater storage in the area, there were 24 nests and 2 nests at the evaporation basins, respectively (Table 9.11). Although results of these limited analyses are not conclusive, they do not appear to support the fundamental hypothesis that the abundance of nesting stilts and avocets at the evaporation basins would decline in response to the availability of adjacent flooded habitat within the Tulare Basin. The 1996–2001 results of nest surveys at the evaporation basins are inconclusive regarding the attraction and use of adjacent freshwater flooded habitat.

It has also been hypothesized that waterbird use of pre-irrigation and seasonally flooded agricultural lands may account for some apparent reduction in Se bioaccumulation in waterbirds. These pre-irrigated agricultural lands are used extensively by wintering and migratory birds (TLDD 1997). The effectiveness of these pre-irrigated and seasonally flooded areas adjacent to the evaporation basins in contributing to dietary dilution of selenium by wintering and migratory waterbirds has not, however, been investigated or documented.

The availability of freshwater habitat in areas adjacent to the evaporation basins contribute to dietary dilution and reduced concentrations of Se for waterbirds nesting and utilizing the evaporation basins. Results of American avocet and stilt egg monitoring at the TLDD South and Hacienda Evaporation Basins after

Table 9.10 Mayfield adjusted stilt and avocet nests at TLDD modified evaporation basins, location, 1994–2004

Basin	1994	1995	1996[a]	1997[b]	1998	1999[b]	2000[b]	2001[b]	2002[b,c]	2003[b]	2004[b]
South	872	755	110	462	168	15	1	107	437	104	1
Hacienda	1,394	497	65	5	50	9	1	1	19	5	8
Combined	2,266	1,252	175	467	218	24	2	108	456	109	9

[a]We assumed avoidance actions to be fully implemented prior to the 1996 nest surveys
[b]No. of nesting birds were insufficient to calculate Mayfield corrected estimates at one or both basins
[c]The observations increase in nesting in 2002 at the South Evaporation Basin resulted from establishment of a tern colony that precluded hazing within the nesting area

Table 9.11 Egg selenium concentrations (μg g^{-1} dry weight) from the TLDD South and Hacienda Evaporation Basins in years with high and low floodwater storage in adjacent areas

			Geometric mean of egg selenium[b]			
			South basin		Hacienda basin	
Spring	Floodwater storage area (hectare)[a]	Number of stilt/avocet nest	Avocet	Stilt	Avocet	Stilt
1996	2,388	175	9.7	–	8.5	–
1997	24,807	467	20.2	14.7	–	–
1998	16,673	218	5.3	–	–	–
1999	0	24	9.5	–	6.5	–
2000	4,411	2	–	–	–	–
2001	2,995	108	–	–	–	–

Source: TLDD unpublished data

[a]Flooded acreage include both on-farm flood storage and flood storage basin

[b]Sample sizes for egg selenium analyses vary among years and (–) denotes no egg collected for analysis

implementation of the avoidance and minimization actions (1996–2004) were examined to determine if egg Se concentrations were lower in years when floodwater habitat was available adjacent to the evaporation basins, as compared to dryer years. The low abundance of nesting avocets and stilts at the South and Hacienda Evaporation Basins in recent years (Fig. 9.1) has precluded obtaining a sufficient sample of waterbird eggs to evaluate trends between wet and dry years. However, one of the highest geometric mean egg Se concentrations observed for avocet and stilts at the South Evaporation Basin occurred in 1997 (Table 9.11), the year with the greatest floodwater storage in the area.

Given these results, the observed decrease in the number of nesting avocets and stilts (Fig. 9.1) is more likely the result of measures implemented to minimize and avoid bird use of the evaporation basins, as opposed to the effects of flooding or other random factors. Results of this case study are consistent with the assumption that effective management for shorebirds requires (1) management of water levels, (2) availability of food resources, (3) managing upland habitat in association with wetlands, and (4) reducing habitat disturbance. The substantial abandonment of the evaporation basins, resulting in nest densities that were sometimes two orders of magnitude below those of the constructed wetland (Table 9.8), indicates the importance of water depth, disturbance by hazing activity, and shoreline slope as basic physical parameters for shorebird nesting habitat.

9.6 Nesting Bird Compensation Habitat Performance

Results of biological monitoring surveys have shown the Compensation Wetland Habitat, using low-Se, saline agricultural drainage water supplies, to be substantially more attractive to nesting shorebirds, and more productive, than originally anticipated. Nest densities at the constructed wetland are several orders of magnitude greater than at the modified evaporation basins. The “attractiveness” of the constructed habitat is further demonstrated by (a) the rapid first-year colonization of the habitat and (b) the rapid decline in use of modified evaporation basins. The timing of transition from use of evaporation basins to the constructed wetland suggests that the specific habitat manipulations at these two sites (depth, shoreline slope) were quickly identified by the target species. The suitability of the constructed wetland habitat was further demonstrated by the high densities of nests in all years, even in wet years when there was substantial alternative habitat available. Predation losses at the compensation habitat were typically very low (<1 %); protection of the nesting habitat may thus account for a portion of the program's success.

From a practical perspective, the very high nest densities observed at the constructed wetland have implications for future analysis of the use of constructed wetlands in compensating for losses similar to those found at the TLDD evaporation basins. Using data from nesting surveys, in combination with information on egg Se concentrations from the South and Hacienda Evaporation Basins, the acreage required to compensate for unavoidable losses at the evaporation basins was calculated using the protocol from Hanson (1993), the USFWS Hen-Wise protocol, the USFWS Egg-Wise protocol (USFWS 1995), and the alternative protocol proposed by Hanson (1995). Results of these calculations indicated, based on monitoring over the period 1996–2004, the calculated acreage required to offset unavoidable loss to American avocet and black-necked stilts at the evaporation basins would be 2 ha (5 acres) using the Hanson (1993) protocol, 1.32 ha (3.25 acres) using the USFWS Egg-Wise protocol, 1 ha (2.5 acres) using the USFWS Hen-Wise protocol, and 2.4 acres using the Hanson (1995) alternative protocol. These current estimates of compensation habitat requirements are substantially (approximately two orders of magnitude) lower than the original 1993 estimate of 84 ha (207 acres). In short, the study demonstrates that highly productive wetlands can be constructed and managed for shorebirds and may be substantially more effective than previously thought at replacing habitats, such as those at evaporation basins.

Nonetheless, there has been variation in the number of nesting birds using the Compensation Habitats. Variability in the seasonal timing and absolute number of nest starts occurring in the wetland reflect the influence of both environmental and biological factors. Seasonal weather patterns, including air temperature, precipitation, and late spring storm activity, appear to influence the seasonal timing of nesting activity. Variation in the regional abundance of American avocets and black-necked stilts, in addition to the availability of other suitable wetland foraging and nesting sites within the region, are also thought to affect the timing and abundance of avocet and stilt nesting at the constructed wetland.

9.7 Winter Waterfowl Performance

Results of three preliminary surveys demonstrate that the 123-ha (305-acre) saline wetland is being used by a wide variety of birds as wintering habitat. The addition of saline wetland habitat provides an increase in the availability of seasonal wetlands within the SJV that benefit both resident and seasonal migrant species. The TLDD plans to continue monitoring at the winter wetland to assess long-term habitat use and variation in use, based on alternative water and planting strategies, as well as in response to variation in precipitation year-to-year and the availability of alternative wetlands.

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Chapter 10
Integrated Irrigation and Drainage Water Management

James E. Ayars and Blaine R. Hanson

10.1 Introduction

In the last decade, a dramatic shift has occurred in the approach to water management in arid and semi-arid irrigated areas. Previously, water management focused on the design and management of irrigation systems, and subsurface drainage provided a means to correct for their deficiencies. As a result of prior design and management, excessive deep percolation losses were routine outcomes of irrigated agriculture, which provided high (excessive) leaching fractions that were captured by the drainage system for disposal. These leaching fractions were then the basis for the drainage system design.

The conventional approach to subsurface drain design was to install the drain laterals as deep as was practical (1.8–2 m), which would result in the largest spacing and minimize the installation cost. These depths of installation were also thought to be required to reduce salinity accumulation in the root zone by upward flow from saline groundwater (Interior 1993). The drainage system was designed to operate continuously and to discharge into a surface water body without any special consideration for environmental consequences. The deep placement of the drains had a secondary impact in that the groundwater flowing to the drains originates from deep within the soil profile and mines salt stored there. The effect was to create a salt load in the drainage water that was in excess of the load needed to maintain salinity in the crop root zone.

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_10,

The need for integrated design and management of irrigation and drainage systems was highlighted as a result of the research conducted during the San Joaquin Valley Drainage Program (San Joaquin Valley Drainage 1990) and other research in arid, irrigated areas of the world. The research demonstrated that source control was critical in reducing drainage flow. Source control in irrigation systems includes the following practices:

- Better management of pre-plant irrigation
- Modified irrigation system layout (reducing field length)
- Changing irrigation system types for better control of applied water (e.g. sprinkle instead of surface)
- Improved irrigation scheduling to improve water use efficiency and reduce loss
- Recycling saline drainage water and *in-situ* use of groundwater
- Both the recycling of saline drainage water and *in-situ* use of groundwater require drainage system management and modification in drainage system layout.

The inclusion of water quality as a design factor for subsurface drainage systems in arid areas represents a major change in water management and in the operation of subsurface drainage systems. Water table management is also being considered in the design of subsurface drainage systems. This has required a reconfiguration of the drainage system laterals and installation of laterals at shallower depths than previously used. Best management practices (BMPs) have been developed for the design and operation of drainage systems (Christen and Ayars 2001). It is a significant step for subsurface drainage systems to be actively managed in areas with significant salinity levels in the soil and groundwater.

This chapter reviews projects and results from research done in drainage-impacted areas in the San Joaquin Valley (SJV), including recommended changes in irrigation and drainage system design and management for systems operating in the SJV and other arid, saline areas.

10.2 Irrigation Research

Subsurface drainage flow can be reduced by (a) improving the existing irrigation system design and management and (b) converting to irrigation systems capable of applying water at high irrigation efficiency. Both approaches require increasing the irrigation efficiency by improving the uniformity of the applied and infiltrated water and their management, e.g. appropriate irrigation set times and frequencies to levels exceeding those of the existing irrigation system. If improving the existing irrigation system management fails to reduce the amount of subsurface drainage flow to levels that can be disposed of by appropriate drainage water disposal methods, then conversion to irrigation systems capable of achieving the desired amount of drainage reduction is necessary. It should be noted that in

Table 10.1 Adjusted crop coefficients (K_c) of cotton for different water table depths and ground water salinity

Water table depth (m)	Base K_c	Adjusted K_c at different groundwater salinity (dS m^{-1})				
		0.3	0.77	7.7	15.4	23.4
1.22	0.06	0.01	0.04	–	0.03	0.05
	0.15	0.11	0.14	–	0.14	0.13
	0.33	0.26	0.26	–	0.28	0.29
	0.59	0.41	0.37	–	0.43	0.50
	0.89	0.51	0.46	–	0.54	0.72
	1.09	0.53	0.48	–	0.58	0.85
1.98	0.06	–	–	0.03	0.06	–
	0.15	–	–	0.14	0.16	–
	0.33	–	–	0.28	0.31	–
	0.59	–	–	0.43	0.47	–
	0.89	–	–	0.54	0.62	–
	1.09	–	–	0.58	0.70	–

The base coefficients represent the no water table condition

addition to subsurface drainage reduction, the improved irrigation systems must also provide sufficient soil salinity control in the root zone to maintain profitable yields. Salinity control involves applying water in excess of the soil water depletion to leach salts out of the root zone. Thus, some subsurface drainage is necessary to leach salts below the root zone, if internal drainage is inadequate to remove deep percolation.

Improved irrigation system management requires improved irrigation scheduling methodologies. An irrigation schedule requires establishing both the time and amount of water application. This is accomplished by multiplying the potential evapotranspiration (ET_{pot}) by a crop coefficient that accounts for the growth stage of the crop, yielding a calculation of crop water use whose estimated values are accumulated to a threshold value of soil water depletion that is used to initiate irrigation. Traditionally, these computations have not accounted for groundwater use by crops. Contributions from the groundwater should change the irrigation schedule by increasing the interval between irrigations and reducing the total seasonal requirement. Failure to include this contribution results in over-irrigation and increased drainage.

Ayars and Hutmacher (1994) developed crop coefficients for cotton, a major SJV crop, that account for the upward flow of shallow groundwater to meet crop water requirements. The upward flow results in soil moisture depletion between irrigations less than predicted by the crop evapotranspiration. These modified coefficients depend on stage of growth, depth to the water table, and the salinity of the shallow groundwater (Table 10.1). The product of the modified crop coefficients and reference crop evapotranspiration is the soil water depletion, not the total crop evapotranspiration, under shallow groundwater conditions. Both timing and depth of application of the irrigation can be established using these modified crop coefficients.

10.2.1 Furrow Irrigation

Furrow irrigation, the most common irrigation method in the drainage problem area of the SJV, is relatively difficult to upgrade and manage efficiently, compared to other irrigation methods, because its performance largely depends on the infiltration rates of water into soil. A main problem with furrow irrigation is the amount of water required and the time it takes for water to reach the downstream end of the field. This time, called the advance time, is highly dependent on the infiltration rate, furrow flow rate, and furrow slope, and less dependent on the furrow surface roughness. It is difficult to apply small amounts of water and not severely deficit-irrigate the field because of the variability of advance time. Management of furrow irrigation systems usually involves a trial and error approach to determine flow rate and irrigation set time because of the spatial and temporal variation in the infiltration rate.

Hanson (1989) evaluated the effectiveness of measures to improve furrow irrigation, using a computer simulation model calibrated by data from field evaluations. Results of the simulations showed that reducing the field length is the most effective measure for reducing deep percolation losses contributing to subsurface drainage flow in furrow-irrigated fields. Compared with the initial field length, decreasing the field length by one-half can reduce subsurface drainage flow by at least 50 % and increase the uniformity of infiltrated water (expressed as the distribution uniformity) by 10–15 %. The irrigation set time must be reduced by 30–40 % for this measure to be effective, because the advance time of the shortened field may be 30–40 % of the time needed for the initial field length. Failure to reduce the set time greatly increases both deep percolation and surface runoff.

A secondary problem with reducing the field length by ½ is the potential increase in surface runoff by 2–4 times, compared with the initial full field length. Controlling the surface runoff may require cutting back irrigation, using tailwater recirculation systems that recover the surface runoff and return it to the head of the field during the irrigation, or on-farm tailwater systems that store the surface runoff for reuse elsewhere on a farm.

Increasing the furrow flow rate is frequently recommended to improve furrow irrigation, since this measure can reduce the time for water to reach the end of the field. However, the study by Hanson (1989) revealed that this measure may have little effect on improving irrigation uniformity and reducing subsurface drainage. The higher furrow inflow rates increased the depth of flow and the wetted area for infiltration. As a result, total infiltrated water increased with inflow rate, which offset the effect of the reduced advance time on uniformity. These results show that the potential effect of reduced furrow length on deep percolation may be difficult to realize when furrow stream size is increased.

In soils that severely crack between irrigations, a reduced field length may have only a slight effect on infiltrated amounts, compared to the original field length, because water flow into the cracks dominates the infiltration process, according to a study by Hanson et al. (1998). The results also showed that irrigation water should be cutoff about 2–3 h after the water reaches the end of the field because, when the water reached the end of the field in the study, water flowing into the cracks rapidly infiltrated to 3–5 ft (0.91–1.52 m) deep. For cracking soils, converting to another irrigation method may be the best feasible solution to substantial subsurface drainage flow reduction.

10.2.2 Sprinkle Irrigation vs. Furrow Irrigation

The performance of sprinkle irrigation systems depends on the hydraulic design and the spatial variability of the applied water due to sprinkler spacing, type, and nozzle size, system pressure, and wind speed and direction. Soil characteristics are considered by designing appropriate system application rates that will not exceed infiltration rates. Because of these design characteristics, sprinkle irrigation can apply small amounts of water reasonably uniformly under low wind conditions; the amount will depend on the management.

Dellavalle Laboratory, Inc. (1995) compared the operation and distribution uniformity of hand- move sprinkle irrigation and improved gated-pipe furrow irrigation of cotton at several locations on the west side of the SJV from 1989 to 1993. The improved furrow irrigation systems consisted of splitting the 800 m long field into three or four sections along its length and using gated pipe to supply water to each section. Surface runoff of an uphill section was allowed to flow beneath the gated pipe of the downhill sections. A tailwater ditch at the end of the field collected the surface runoff.

At one location, the focus was on the pre-plant irrigation, considered to be a major source of subsurface drainage water. Sprinkle irrigation was used on the east half of the demonstration field for the pre-plant irrigation, while improved furrow irrigation was used on the west half. The improved furrow irrigation system consisted of dividing the field length (800 m; 2,625 ft) into three sections of equal length (268 m; 879 ft) and installing gated pipe at the head of each section. The improved furrow irrigation system was used for crop irrigations for the entire field.

Management of the improved furrow irrigation system involved irrigating the lower one third of the field first, until water reached the end of the section. The irrigation water to that section was then terminated and irrigation was started on the middle section. When the water reached the end of that section, the irrigation of that section was terminated, and irrigation was then started on the first section (at the head of the field). The pre-plant irrigation required about 5 h for the water to advance to the end of each section. However, a total irrigation time of about 30 h was needed to adequately wet the seedbed to the soil moisture level needed for good germination.

Pre-plant irrigation amounts of applied water were 102 and 226 mm (4–8.9 in.) for sprinkle and improved furrow irrigation, respectively. However, considerable under-irrigation occurred for the sprinkler systems. It was estimated that 229 mm (9 in.) of water was required for soil moisture replenishment of the profile.

The seasonal crop irrigation amounts were 460 mm (18 in.) for both sides of the field. While a smaller amount of water was applied by the sprinkle pre-plant irrigation, measurements of furrow infiltration rates during the first crop irrigation showed infiltration rates on the east half of the field (pre-plant sprinkle irrigation) to be nearly twice those of the west half (pre-plant furrow irrigation), thus raising the potential of more infiltrated water during the first crop irrigation for pre-plant sprinkle irrigation compared to pre-plant furrow irrigation.

At a second location, sprinkle irrigation was used for the 1991 pre-plant irrigation on the east half and improved furrow irrigation on the west half. The improved furrow irrigation system was used for the seasonal crop irrigations; however, sprinkle irrigation was used for germination and stand establishment. The improved furrow system consisted of dividing the field into three sections (268 m; 879 ft long) with gated pipe supplying the water to each section.

About 238 mm (9.4 in.) of water were applied with sprinkle irrigation during the pre-plant irrigation of the east half of the field, while 125 mm (4.9 in.) were applied with improved furrow irrigation. The difference was attributed to the timing of the pre-plant irrigations. The sprinkle irrigation occurred in early February when the soil moisture depletion of the top 1.5 m (4.9 ft) of the soil profile was 178–229 mm (7–9 in.). The pre-plant furrow irrigation occurred in the middle of March, after about 71 mm of rainfall. The timing of the pre-plant irrigation was a management decision made by the grower. The seasonal amount, applied during the crop season, was 409 mm (16 in.) for both sides of the field.

In the following year (1992), sprinkle irrigation only was used for the pre-plant irrigation on both sides of the field, while improved furrow irrigation with four lines of gated pipe was used for the crop irrigations (except germination and stand establishment). Pre-plant irrigation amounts were 58 mm (2.28 in.) and 122 mm (4.80 in.) for east and west halves, respectively. The irrigation time of the pre-plant irrigation of the east half of the field, which occurred in February, was 24 h, while that of the west half was 12 h (occurred in March). The irrigation time was reduced because rainfall between the two irrigation events decreased the soil water deficit from 94 mm (3.7 in.) for the February irrigation to 53 mm (4.8 in.) for the March irrigation. The seasonal irrigation amount was 533 mm (21 in.) for both sides of the field.

In 1993, no pre-plant irrigation occurred because of rainfall. The seasonal crop irrigations consisted of improved furrow irrigation (east half – 268 m [798 ft] furrow lengths) and solid set sprinkler irrigation (west half). Sprinkle irrigation was used for germination. The seasonal amount of the crop irrigations was 460 mm (16 in.). These results show that sprinkle irrigation can apply small amounts of irrigation water and can greatly reduce drainage below the root zone. No conclusions can be made about the effect of reduced furrow length because the improved furrow irrigation systems were not compared with the historical furrow irrigation systems. However, the timing of the pre-plant irrigation, relative to winter and spring rainfall, can affect the amount of applied water and thus drainage, provided the irrigator accounts for rainfall in the management of the irrigation system.

Table 10.2 Comparison of yield, applied water and net returns for subsurface drip, improved furrow, surge irrigation, and historic furrow irrigation systems

Irrigation method	Cotton yield (kg hectare^{-1})	Applied water (mm)	Net returns ($ hectare^{-1})
Drip	1,590	556	504
Improved furrow	1,430	612	990
Surge	1,430	622	990
Traditional furrow	1,430	660	1,020

10.2.3 Drip Irrigation

10.2.3.1 Field Experiments

One option for improving irrigation water management is to convert from furrow or sprinkle irrigation to drip irrigation. Drip irrigation has the potential to apply water both precisely and uniformly at a high irrigation frequency, compared with furrow and sprinkle irrigation, resulting in the potential to reduce subsurface drainage, control soil salinity, and increase yield. The ability to do this is not only governed by the technology, but also by design, installation, operation, and maintenance. One disadvantage of drip irrigation is its installation cost, which based on grower experience, could range from $1,480 to $2,470 ha^{-1} ($601–$1,000 acre^{-1}). Use of drip irrigation requires improved management capability, since the crop water requirement is being met by the applied irrigation water and not necessarily by stored soil water. If the drip irrigation system breaks down or its management is poor, there is potential for significant crop loss

Fulton et al. (1991) compared a subsurface drip irrigation system, an upgraded furrow system, and a surge irrigation system with the traditional furrow system on clay loam. Crop type was cotton. The traditional furrow length was 720 m (2,362 ft). A 360 m (1,181 ft) furrow length was used for the pre-plant irrigation for both the improved and surge systems; a 720 m (2,362 ft) length was used thereafter. Drip tubing was buried about 0.45 m (1.48 ft) deep with a lateral spacing of about 1 m (3.3 ft) and lateral lengths of about 182 m (597 ft).

Cotton yield was about 12 % higher for the drip system, compared with the furrow systems (Table 10.2). No differences in yield occurred among the furrow systems. Water applications were about 56, 66 and 104 mm (2.2, 2.6, and 4.0 in.) more for the improved furrow, the surge system, and the traditional furrow system, respectively, compared to the drip system. Surface runoff was assumed to be used beneficially. Net returns were the largest for the traditional furrow, followed by the improved furrow and surge systems. Net returns for the drip system were about one-half those of the furrow systems.

Styles et al. (1997) compared subsurface drip irrigation and furrow irrigation of cotton on a clay loam soil from 1989 to 1993. In 1989, no improvements were made to the furrow system (field length of about 360 m; 1,181 ft), while in 1990, an

Table 10.3 Comparison of yield, applied water and net return of subsurface drip, improved furrow, and historic furrow irrigation methods

Irrigation method	Cotton yield (kg hectare^{-1})	Applied water (mm)	Net returns (\$ hectare^{-1})
Drip	1,455	564	266
Improved furrow	1,250	597	205
Traditional furrow	1,250	658	239

Values are averages over the duration of the project

improved furrow system with a furrow length of 180 m (590 ft) was used along with the historic furrow system (360 m; 1,181 ft furrow length). Drip tubing was buried 0.45 m (1.48 ft) deep with a lateral spacing of 2.03 m (6.7 ft). Hand-move sprinkle irrigation was used for stand establishment of the drip system.

Average cotton lint yields of the drip system were about 16 % higher than the furrow yields (Table 10.3). Less water was used by the drip system, compared to the furrow systems, and average net returns were higher for the drip system for the project duration.

A study (Ayars et al. 2001) was conducted using subsurface drip irrigation (SDI) and furrow irrigation on tomato and cotton crops grown on two fields without active drainage systems. Five types of drip tubing were used. The tubing was installed at a depth 45 cm (17.7 in.) with a spacing of 1.6 m (5.25 ft) on the tomatoes and 2 m (6.56 ft) on the cotton. The furrow irrigation used 400 m (1,312 ft) runs for the cotton and 260 m (853 ft) runs for the tomatoes. The water table depth was monitored using observation wells. The total tomato yield was 131 Mg ha^{-1} (58.4 t acre^{-1}) using the SDI, compared to 110 Mg ha^{-1} with the furrow irrigation. The cotton yield increased from 1,390 to 2,040 kg ha^{-1} (1,555–2,282 lb acre^{-1}) over the 3 years of the study, while the yield in the furrow ranged from 1,460 to 1,540 kg ha^{-1}. The depth to the water table decreased with time under the tomatoes, until the furrow irrigation ceased. At that time, the depth stabilized and began to increase. This pattern was not observed under the cotton crop. The major differences were the frequency and depth of irrigation between the two crops. The tomato was irrigated more frequently than the cotton, and the total application was greater, which resulted in increased deep percolation losses under the tomato. Also, cotton used considerable water from the shallow groundwater. The water table recession after the tomato crop was adequate to permit pre-plant irrigation, which controlled the root zone salinity.

One problem encountered with the SDI was damage to the tubing, both from rodents and from tillage implements. The rodent damage was minimal on the heavy-wall drip tubing and was heaviest on the thin-wall drip tubing. The problems with the mechanical damage occurred as a result of the farm changing the row spacing for the tomatoes between crops: The tubing was no longer being placed under the crop row but, instead, in the middle, between crop rows. This shift also impacted salt development; salinity was being moved into the root zone. Normally, salts would have been leached from the root zone (Fig. 10.1). The chloride data in Fig. 10.1 are representative of the distribution of salinity in the profile under the individual beds.

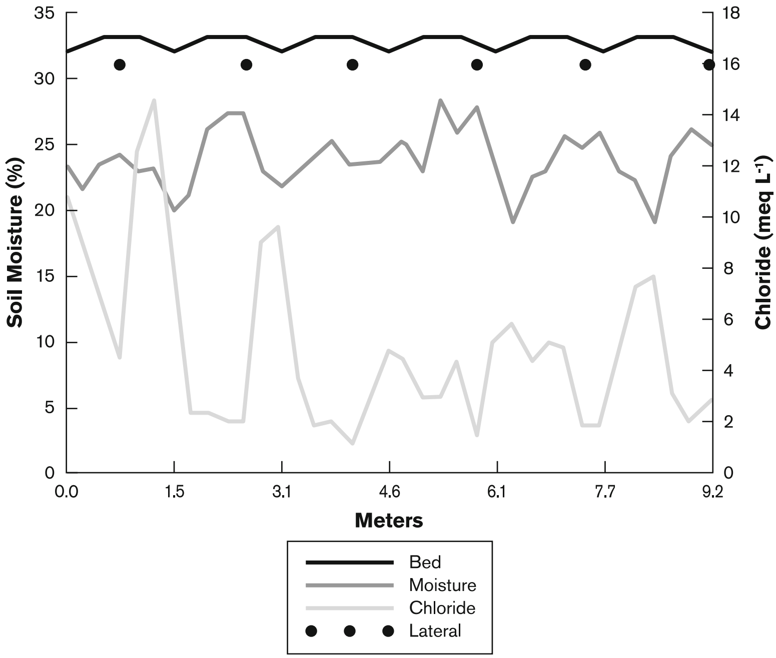

Fig. 10.1 Distribution of chloride ion (Cl) under subsurface drip irrigation system at Britz farms

Experiments in three commercial fields of processing tomatoes compared SDI and sprinkle irrigation under saline, shallow groundwater conditions (Hanson and May 2004). Tomatoes are a high cash value crop compared to cotton, but they are classified as moderately sensitive to soil salinity with a Maas-Hoffman threshold (Maas 1990) electrical conductivity (ECe) of 2.5 dS m^{-1} compared to 7.7 dS m^{-1} for cotton. The threshold ECe is the maximum average root zone salinity at which yield is not reduced. Results showed SDI of processing tomatoes to be highly profitable, compared to sprinkle irrigation, under saline, shallow groundwater conditions. High profitability occurred even with saline water table depths of 45–60 cm (17.7–23.6 in.). The average yield of SDI was 89.1 Mg ha^{-1} vs. 73.5 Mg ha^{-1} (99.7 lb $acre^{-1}$ vs. 82.2 lb $acre^{-1}$) under sprinkle irrigation. Yield of SDI was unaffected by the range of soil salinity in these fields (discussed later). The average increase in profit as a result of converting from sprinkle to SDI was $1,195 ha^{-1} ($484 $acre^{-1}$).

Soil salinity around drip lines was found to depend on the depth to the groundwater, the salinity of the shallow groundwater and irrigation water, and the amount of applied water. Groundwater salinity at these fields ranged from 8 to 11 dS m^{-1} while irrigation water salinity (EC_i) ranged from 0.34 to 1.1 dS m^{-1}. For a water table depth of about 2.0 m (6.6 ft) during the crop season, relatively uniform soil salinity was found around the drip lines, with values smaller than the threshold ECe, regardless of the groundwater salinity (8–11 dS m^{-1}) (Fig. 10.2a). For water table depths less than about 1 m, soil salinity varied considerably around drip lines, with the smallest levels near the drip line for the same EC_i (Fig. 10.2b). Soil salinity

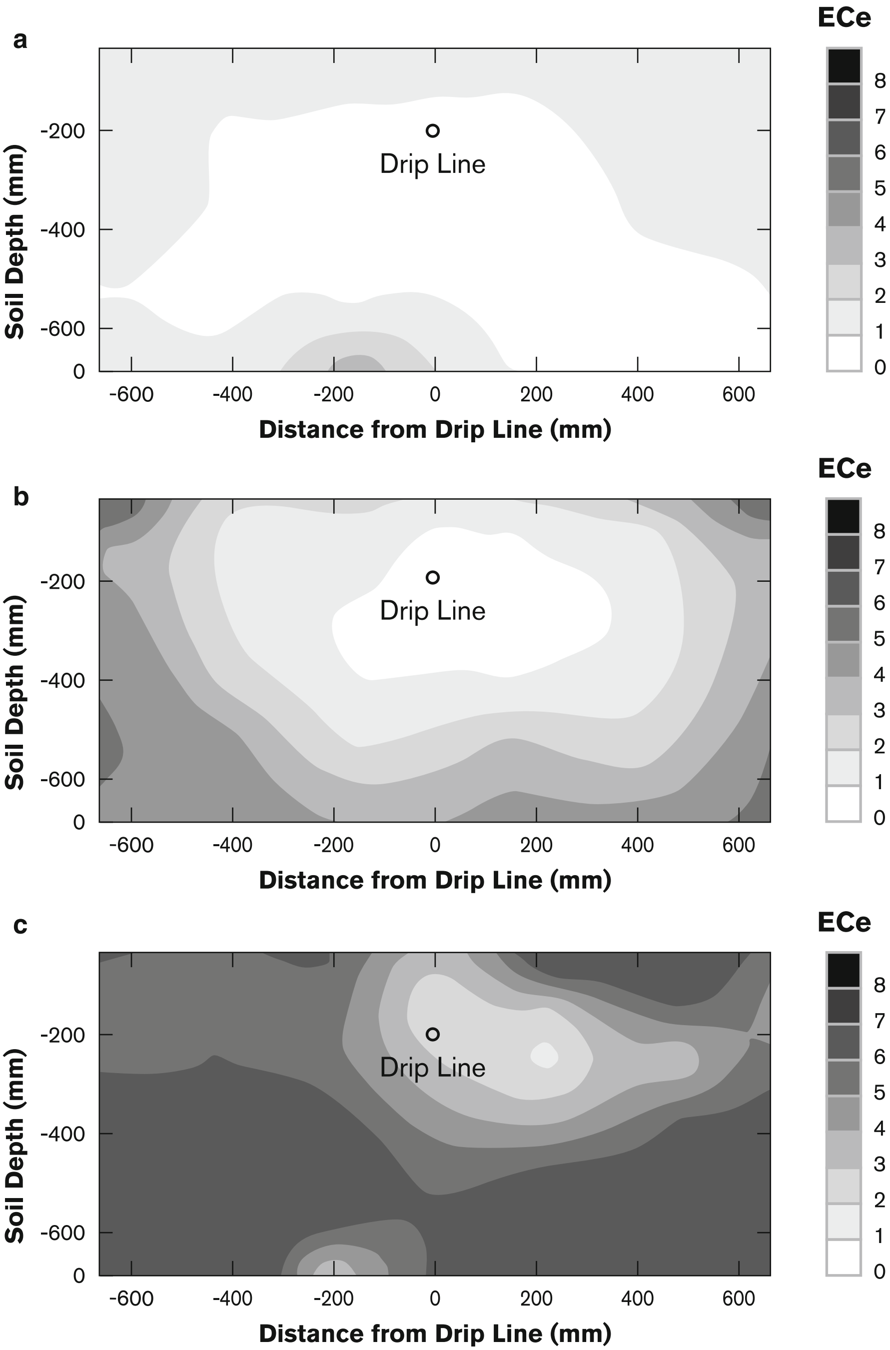

Fig. 10.2 Soil profile salinity around drip lines measured as dS m^{-1} of ECe for (**a**) average water table depth = 2 m, EC_{iw} = 0.3 dS m^{-1}, and EC_{gw} = 8–11 dS m^{-1}; (**b**) water table depth = 0.61–1 m, EC_{iw} = 0.3 dS m^{-1}, and EC_{gw} = 5–7 dS m^{-1}; and (**c**) water table depth = 0.61 and 1 m, EC_{iw} = 1.1 dS m^{-1}, and EC_{gw} = 9–16 dS m^{-1}

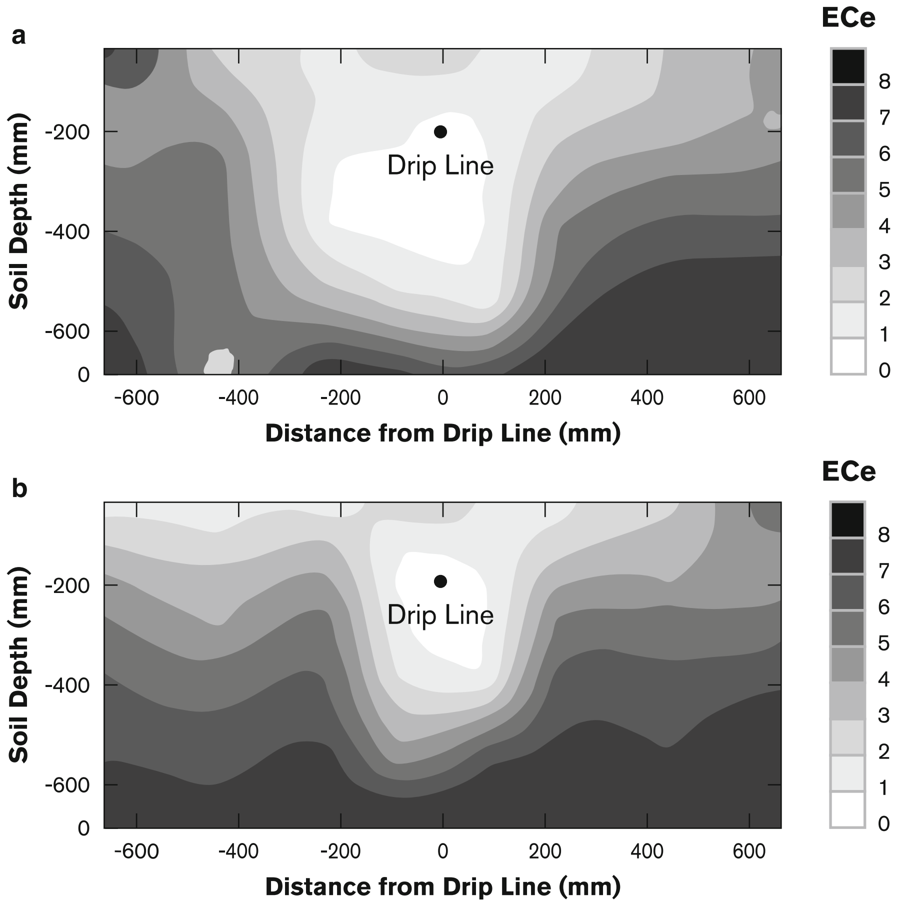

Fig. 10.3 Soil profile salinity around the drip line measured as dS m^{-1} of ECe that respectively received (**a**) 371 mm, comparable to seasonal ET for processing tomatoes and (**b**) 211 mm of irrigation water. $EC_{iw} = 0.52$ dS m^{-1} and $EC_{gw} = 8\text{–}11$ dS m^{-1}

around the drip line increased, as the EC_i increased (Fig. 10.2c). The larger the amount of applied water, the larger was the zone of low-salt soil near the drip line (Fig. 10.3).

At a fourth commercial field, small plot experiments were conducted to determine the effect of the applied water amount on the yield of processing tomatoes and cotton under very shallow, saline groundwater conditions (0.45–0.60 m; 1.5–2.0 ft) (Hanson et al. 2006). Tomato yield decreased from 94.2 Mg ha^{-1} (considered to be a high yield) to 76 Mg ha^{-1} as the applied water decreased from 590 mm (23.2 in.) (about equal to the seasonal evapotranspiration of tomatoes) to 396 mm (15.6 in.). Cotton yield was not affected by the amount of applied water for application amounts greater than 50 % of the ET_{pot}. Other studies have demonstrated that cotton can use significant (>50 %) quantities of saline groundwater (EC < 15 dS m^{-1}) *in-situ* (Hutmacher et al. 1996; Ayars et al. 2006a).

Table 10.4 Seasonal applied water and evapotranspiration, and field-wide leaching fractions calculated from a water balance for the four commercial tomato sites (BR, DI, DE, and BR2)

Site	Years	Applied water (cm)	Evaporation (cm)	Leaching fraction (%)
BR	1999	410	520	0
	2000	430	540	0
	2001	520	580	0
DI	1999	560	640	0
	2000	740	640	13.1
	2001	580	680	0
DE	2000	730	610	13.6
	2001	560	590	0
BR2	2002	590	620	0

At these sites (processing tomato), there was little or no response of the water table to drip irrigation, except at one site in 1 year, when over-irrigation occurred early in the crop season. Reduced water applications resulted in a declining water table. Ayars (2003) found similar results in a study comparing SDI to surface irrigation methods when the water table response was linked to over-irrigation by a furrow irrigation system and the depth to water table increased after furrow irrigation ceased.

As discussed earlier, a key to the profitability of drip irrigation of tomatoes in the SJV's salt-affected soils is salinity control. Differences between seasonal applied water and evapotranspiration of these sites (water balance approach to calculating the amount of leaching) generally showed little or no leaching most of the time (Table 10.4), which raises questions about the long-term feasibility of drip irrigation due to inadequate salinity control. However, the soil salinity patterns around drip lines showed that considerable, localized leaching occurred around drip lines. Estimating the amount of actual leaching under drip irrigation is difficult because root density, soil salinity, and soil water content all vary with distance and depth from the drip line. Based on these results, it was concluded that applying an amount of water nearly equal to the ET_{pot} provided sufficient actual leaching near the drip line, an area where root density and, thus, root water uptake usually is the highest under drip irrigation.

10.2.3.2 Computer Simulation: Subsurface Drip Irrigation of Processing Tomatoes

Because of the difficulty in estimating actual leaching fractions, computer simulation (HYDRUS-2D) was used to describe water and salt flux with SDI under saline, shallow groundwater conditions and to estimate leaching fractions (Hanson et al. 2008). The HYDRUS-2D model simulated the movement of water and salt in soil under SDI and determined the amount of water draining below the root zone. Simulations were conducted for water table depths of 50 cm (19.7 in.) with daily irrigation and 100 cm (35.4 in.) with two irrigations per week, irrigation water salinities (EC_i) of 0.3, 1.0, and 2.0 dS m^{-1}, and applied water amounts of 60 % (EC_i = 0.3 dS m^{-1} only), 80, 100, and 115 % of the $ET_{pot.}$ Depth of the drip line was 20 cm (7.87 in.). The initial soil salinity conditions were based on field measurements of soil salinity in the spring, prior to any drip irrigation.

Results of these simulations showed that reclamation of the soil near the drip line was rapid (Fig. 10.2), but larger amounts of water per irrigation applied less frequently increased the rate of reclamation compared to small amounts applied daily. As time progressed, the volume of reclaimed soil increased, with most of the reclamation occurring below the drip line and salt accumulating above the drip line near the soil surface. The larger the amount of applied water, the larger the volume of reclaimed soil below the drip line, but the amount of applied water had little effect on the volume of reclaimed soil above the drip line. Even a water application of 80 % (considered to be deficit irrigation) resulted in considerable reclamation near the drip line. The salinity near the drip line increased as the EC_i increased.

Actual or localized leaching fractions ranged from 7.7 % (60 % water application) to 30.5 % (115 % water application). The leaching fraction was 24.5 % for the 100 % water application. Even for applications considered to result in deficit irrigation, drainage below the root zone occurred, with a leaching fraction of 17 % for a water application equal to 80 % of the ET_{pot}. This behavior is due to the wetting patterns that occur under drip irrigation. These data, coupled with the measured soil salinity data, indicate that the water balance approach (applied water minus evapotranspiration) for measuring leaching fractions is inappropriate for drip irrigation because of the limited area being wetted by the drip system.

A common assumption is that an amount of applied water equal to 100 % ET_{pot} will result in 100 % irrigation efficiency for drip irrigation with little drainage below the root zone. The computer simulation study showed this assumption is not true. Because of the spatially varying soil water wetting around drip lines, the irrigation efficiency, defined as the ratio of the cumulative root water uptake to the applied water, was 74.6 and 69.7 % for the 100 cm (39.4 in.) and 50 cm (19.7 in.) water table scenarios, respectively. Very high irrigation efficiencies occurred only under severe deficit irrigation conditions. However, because of high frequency irrigation, the volume of drainage per irrigation is small, and the drainage is distributed evenly over the irrigation season. As a result of this behavior, the natural subsurface drainage in the previously-discussed commercial fields appeared to be sufficient to prevent groundwater intrusion into the root zone.

10.3 Drainage Research

The problems at Kesterson Reservoir were a result of evaporation of saline drainage water containing selenium (Se) as well as salt (San Joaquin Valley Drainage 1990) in ponds located in a wildlife refuge. An extensive research program was established to provide solutions for environmentally sound disposal of saline drainage water. One research component was evaluation of the potential for using

Table 10.5 Rainfall and applied water in furrow irrigated plots, F1 and F2

Year	Crop	Rainfall (mm)		Applied water (mm)			
		Between planting and harvest	Planting to planting (total)[a]	Plot F1		Plot F2	
				NS[b]	S[b]	NS[b]	S[b]
1982	Cotton	42	287	276	0	502	0
1983	Cotton	64	126	302	0	302	0
1984	Cotton	29	128	715	0	725	0
1984/1985	Wheat	60	64	532	0	535	0
1985/1986	Sugar beet	245	349	836	0	114	720
1987	Cotton	68	68	770	0	270	515
Total (from 1982 planting through 1987 harvest)		508	1,022	3,631	0	2,448	1235

[a]Total rainfall received from planting time for the year to planting time of the subsequent crop (in either the same or the next calendar year). The 1987 data are for period of planting to harvest
[b]For low-salinity water (NS), $EC_w = 0.3$ to 0.5 dS m^{-1}. For saline water (S), $EC_w =$ 3.5–3.8 dS m^{-1}

saline drainage water for supplemental irrigation as part of the overall drainage water disposal plan. Additional studies evaluated the effect of water table management on the total drainage flow. The following sections discuss projects that were part of the aforementioned research.

10.3.1 Water Table Control: Drainage Water Reuse

A 5-year field study (Ayars et al. 1993) was conducted on Murrieta Farms located on the west side of the SJV to evaluate the potential and problems associated with the use of saline drainage water for irrigation. The study compared surface drip application (T1, T2) of saline drainage water to furrow irrigation (F1, F2) using run lengths of 400 m (1,312 ft). The saline drainage water was taken from the existing drainage system and stored in a tail water pond for application. The crop rotation included cotton, cotton, wheat, sugar beet and cotton and the Maas-Hoffman thresholds were 7.7 dS m^{-1} for cotton and 6 dS m^{-1} for wheat and 7 dS m^{-1} for sugar beet. The EC of the saline water was 7 dS m^{-1} with a boron (B) content of 5 mg L^{-1} while the low-salinity water had an EC of 0.4 dS m^{-1} and no B. The drip irrigated crops were pre-plant irrigated with low-salinity water and followed with saline water for the entire growing season. The exception was the wheat crop, which was irrigated only with low-salinity water. The results are summarized in Tables 10.5 and 10.6. The water balance data show that significant quantities of saline (S) water were used in crop production, which reduced the total volume for disposal. However, the B data in the soil profile at the end of the experiment indicated that if reclamation was needed to restore B levels to the initial values, it would take an amount of low-salinity water equal to the saline water used in irrigation. This would not be a net saving of water.

Table 10.6 Rainfall and applied water in drip plots, T1 and T2

		Rainfall (mm)		Applied water (mm)			
				Plot T1		Plot T2	
Year	Crop	Between planting and harvest	Planting to planting (total)[a]	NS[b]	S[b]	NS[b]	S[b]
1982	Cotton	42	287	580	0	580	0
1983	Cotton	64	126	175	200	175	200
1984	Cotton	29	128	150	336	150	547
1984/1985	Wheat	60	64	526	0	526	0
1985/1986	Sugar beet	245	349	114	666	114	973
1987	Cotton	68	68	155	573	155	749
Total (from 1982 planting through 1987 harvest)		508	1,022	1,700	1,775	1,700	2,469

[a]Total rainfall received from planting time for the year to planting time of the subsequent crop (in either the same or the next calendar year). The 1987 data are for period of planting to harvest
[b]For low-salinity water (NS), $EC_w = 0.3–0.5$ dS m^{-1}. For saline water (S), $EC_w = 7.2–7.9$ dS m^{-1}

Table 10.7 Crop yields of non saline water and non saline-saline water combination furrow irrigated fields F1 and F2 respectively and drip irrigated fields T1 and T2, respectively

			Yields (Mg hectare^{-1})			
Year	Crop	Commodity	Plot F1	Plot F2	Plot T1	Plot T2
1982	Cotton	Lint	1.6	1.4	1.6	1.6
1983	Cotton	Lint	0.9	1.0	1.1	1.1
1984	Cotton	Lint	1.6	1.6	1.8	1.8
1984/1985	Wheat	Wheat grain	7.0	7.0	5.7	5.2
1985/1985	Sugar beet	Sugar	10.5	10.8	10.8	10.2
1987	Cotton	Lint	n.d.[a]	1.5	1.5	1.6

[a]n.d. to indicate no data

The yield data (Table 10.7) show that, with the exception of wheat, the yields were not adversely impacted by the use of saline water. The residual salt in the soil profile reduced the wheat yield for all irrigation treatments. The nitrate (40 mg L^{-1}) in the drainage water increased nitrogen levels in the sugar of the sugar beet, which reduced overall quality even though yields were not reduced. The cotton yields were comparable to the yields in the furrow-irrigated crops.

This study also demonstrated that cotton uses significant quantities of water from shallow groundwater. The in-line water meters that were being used to measure drainage flow in each lateral acted as a passive control on the water table levels upstream from the meters. This provided increased time for in-situ crop water use (Ayars and Schoneman 1984, 1986).

A study was conducted on Cilker farms (Ayars et al. 2006b) to evaluate the design and operation of a subsurface drainage control system. The drain laterals

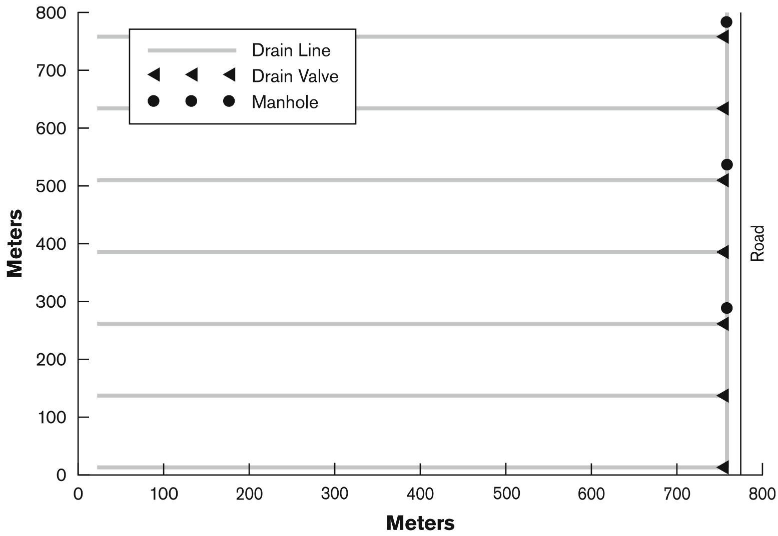

Fig. 10.4 Field layout of water control structures (Cilker Farms study, Ayars et al. 2006b)

had been installed perpendicular to the surface grade, which made this study possible. Individual valves were installed on each lateral and additional structures were installed along the sub-main to control several laterals (Fig. 10.4). The data plotted in Fig. 10.5 show that use of the individual valves made it possible to control the water table position across the field (Fig. 10.5a). Opening the valves increased the depth to the water table across the field within a week (Fig. 10.5b). Root zone salinity accumulation was not a problem at this site (Wu et al. 2001). In this study and the previous study, root zone salinity was managed effectively through a combination of rainfall and pre-plant irrigation. Pre-plant irrigation in excess of 150 mm (5.9 in.) was not required. Previous studies had demonstrated that 300 mm (12 in.) was typically used for pre-plant irrigation and this resulted in excessive deep percolation losses.

10.4 Discussion

10.4.1 Irrigation Water Management

10.4.1.1 Drainage and Salinity Considerations

The primary source of the saline shallow groundwater is subsurface drainage from irrigated land; there is also some contribution due to lateral flow from upslope sources. Pre-plant furrow irrigation has been found to be a major source of subsurface drainage. Contributing factors include the relatively high infiltration

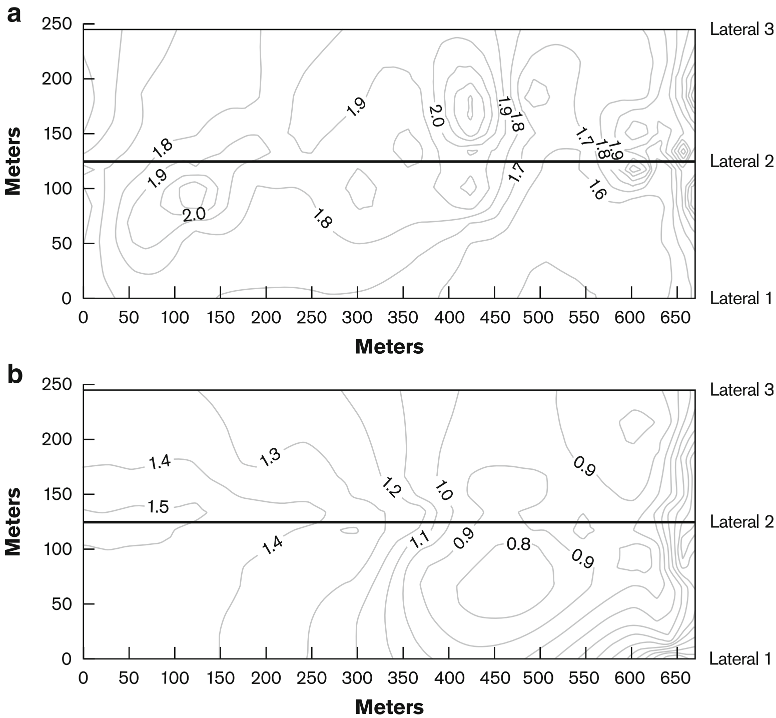

Fig. 10.5 Depth to the water table on May 2, 1994 (**a**) upon control valves of drainage system being opened on April 21 (**b**)

rates that occur at the time of pre-plant irrigation and the 800 m (2,625 ft) long furrow lengths that have been used historically along the west side of the SJV. During the crop season, deep percolation losses are reduced because of the increase in soil water storage due to plant use and also a reduction in infiltration rates.

Improved irrigation practices should be implemented to reduce subsurface drainage. However, salinity will be a limiting factor on the feasible amount of drainage reduction. Some minimum amount of leaching must occur to prevent adverse levels of soil salinity from accumulating in the root zone. Salinity control consists of infiltrating an amount of water in excess of the soil water depletion to leach or transport salts below the root zone. The need for salinity control means that there is a lower limit on the amount of drainage reduction, without incurring a yield decrease due to soil salination.

Pre-plant irrigation plays a major role in controlling soil salinity in the drainage problem areas for furrow and sprinkle irrigation. Infiltration rates during pre-plant irrigations generally are high, thus allowing sufficient water to flow through the root zone to leach salts. The soils on the west side of the SJV tend to be cracking clays, which maintain the infiltration rate into the irrigation season. However, during the seasonal irrigations, total deep percolation losses are reduced, due to increased soil

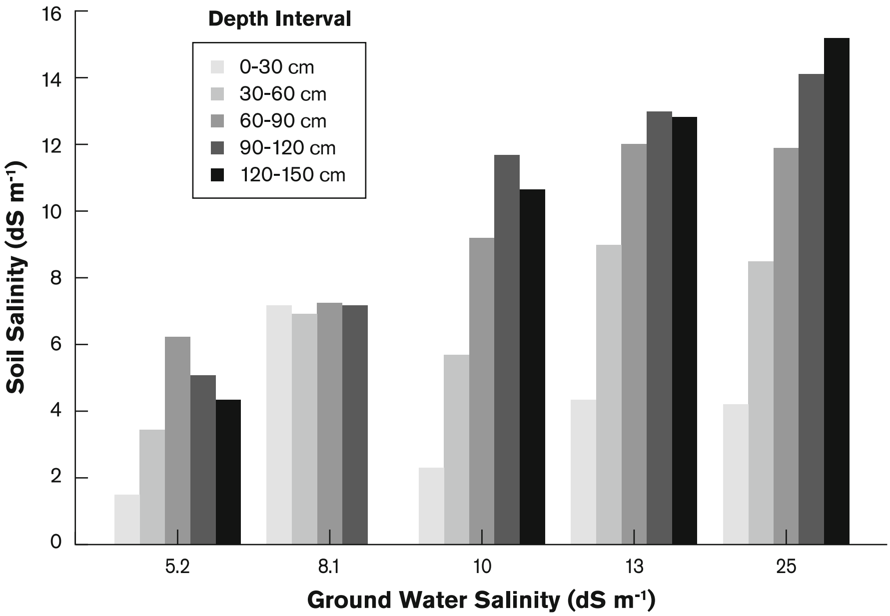

Fig. 10.6 Influences of ground water quality on soil salinity of unsaturated zone above the water table

water storage, and little or no leaching may occur during those irrigations. In areas with shallow groundwater, its upward flow is at a maximum late in the growing season and salt may be transported into the root zone.

Grimes et al. (1984) studied the effect of pre-plant irrigation (or lack of) on soil salinity for a 2-year period. Salt accumulated in the root zone during the growing season from upward flow of the shallow groundwater, causing higher fall soil salinity levels compared to spring levels. The pre-plant irrigation reduced the fall levels down to the spring levels each year. Where no pre-plant irrigation occurred, soil salinity levels in spring remained high. Ayars et al. (1993) also found similar patterns of increased salinity during the growing season, which returned to previous levels with rainfall and pre-plant irrigation between crops.

The distribution and concentration of salts in the soil profile depend on the following:

- Irrigation method
- Amount, timing, and ECi, of the applied irrigation water
- Salinity of the groundwater
- Depth to the water table
- Soil type.

Under sprinkle, flood, and furrow irrigation, soil salinity of a leached soil is the smallest near the ground surface, reflecting the salinity of the irrigation water. At the deeper depths, soil salinity reflects the salinity and depth of the shallow groundwater. Figure 10.6 shows that soil salinity at the deeper depths increased as the salinity of the groundwater increased, but salinity levels near the surface were similar, regardless of the groundwater salinity with one exception: Under drip irrigation, soil salinity is the smallest near the drip line (discussed earlier), but increases with distance and depth (Fig. 10.2).

10.4.1.2 Irrigation Water Management Considerations

One essential aspect of improved irrigation is its management, particularly the timing of irrigation and the amount of applied water. Normally, under deep groundwater conditions, an allowable depletion of soil water, coupled with estimates of the potential crop evapotranspiration (ET) or water use, determines the time of irrigation and the amount of water to be applied. Under shallow groundwater conditions, upward flow of groundwater into the root zone complicates irrigation water management. For a given ET rate and soil type, soil water depletions may be smaller under shallow groundwater conditions than for deep groundwater conditions, due to crop use of the shallow groundwater, and thus, irrigation water management must adjust to account for this use. For some crops, such as cotton, increased use of shallow groundwater is encouraged to reduce subsurface drainage. However, for crops such as processing tomatoes, use of the saline, shallow groundwater is not recommended because of potential yield reductions due to soil salinity. Because of the upward flow contribution, the water balance approach to irrigation water management is inappropriate under shallow groundwater conditions.

A common approach used in California for estimating potential ET is to multiply a reference crop ET (California Irrigation Management Information System [CIMIS]) by a crop coefficient, which depends on crop type and stage of growth. The crop coefficients of cotton (Table 10.1) are recommended for estimating the amount of water to be applied under saline, shallow groundwater conditions since they account for upward flow from the water table. These crop coefficients might be used on other salt-tolerant crops, such as sugar beet.

Soil moisture sensors can help monitor the irrigation water management for drip-irrigated processing tomatoes in a saline soil where the water table depth fluctuates between 45 and 60 cm (1.5 and 2.0 ft). For this situation, careful irrigation water management is needed to prevent excessive wetting of the very shallow root zone. Soil water potential at 0.15 and 0.30 m (0.49 and 0.98 ft) depths ranged from 10 to 60 kPa, reflecting irrigation and soil water uptake, which suggest adequate irrigation water management. When over-irrigation took place, little change in soil water potential occurred at those depths. At depths of 0.46 and 0.61 m (1.5 and 2.0 ft), soil water potential was smaller than −30 kPa, indicating a nearly saturated soil.

10.4.1.3 Irrigation System Considerations

The preferred irrigation method for pre-plant irrigation is sprinkle irrigation. This irrigation method can apply small amounts of water throughout a field, thus greatly reducing percolation below the root zone, yet providing adequate leaching of salt. Hand-move sprinkle irrigation systems are commonly used along the west side of the SJV with sprinkler and lateral spacing ranging from 10.7 to 13.7 m (30 to 45 ft).

Continuous-move sprinkler machines, linear-move and center-pivot systems, have a potential for high irrigation efficiency and thus a large reduction in subsurface drainage. However, they have been tried many times in the Central Valley of California, but are not used because of technical, management, and economic problems. For example, center pivot systems are moderate capital cost, low-labor irrigation systems commonly used in the Midwest. Experience has shown that they are inappropriate for the soils along the west side of the SJV because center pivot application rates needed to meet crop demand at the peak of the growing season greatly exceed infiltration rates of these soils, resulting in excessive surface runoff.

Reducing the furrow length is an effective practice for reducing subsurface drainage, if irrigation set times are adjusted because of the smaller advance times to the end of the furrows. Field demonstrations have shown that dividing an 800 m (2,625 ft) long field into three 267 m (876 ft) sections is a practical approach in terms of effectiveness, labor costs, and capital costs. The length selected will depend on the soil type, the irrigation system, and the crop. Studies have shown that in the presence of shallow groundwater, cotton can be grown effectively with only two 400 m sections, while three 267 m (876 ft) sections are a better fit for tomatoes in 800 m long fields with heavy (clay loam, silty clay loam) soils. A common strategy used on the west side of the SJV is to use sprinkle irrigation for the pre-plant irrigation followed by furrow irrigation with small furrow lengths for the crop irrigations.

Drip irrigation has a high potential for reducing subsurface drainage and providing root zone salinity control because of its ability to apply water uniformly throughout a field at a high frequency. Drip irrigation system design involves a tradeoff between uniformity of applied water and system costs. Short drip lines may result in high uniformity of applied water, but capital costs will be high compared to longer drip lines. A 19 or 21 mm (0.75 or 0.83 in.) diameter drip line is commonly used for drip line lengths of about 365 m (1,198 ft). Other design considerations include emitter spacing and emitter discharge rate, and whether to use surface drip irrigation or subsurface drip irrigation. Surface drip irrigation is normally used for vegetable crops, while subsurface drip irrigation is used for processing tomatoes and cotton in areas with salt-affected soil. Improvements in both the drip equipment and the machines needed to install and retrieve drip irrigation have made drip irrigation a viable option for use on the west side of the SJV.

Are subsurface drainage systems and drainage water disposal methods needed under drip irrigation? No subsurface drainage systems were used in the commercial fields evaluated by Hanson and May (2004) and Hanson et al. (2006). Water table depths ranged from 0.45 to 2 m (1.5 to 6.6 ft). At all sites, groundwater salinity was high. Yet, no trend in yield response with water table depth or soil salinity was found over the study period. Subsurface drip irrigation continues to be used at this time at these sites. Also, substantial localized leaching occurred around the drip line at all sites, regardless of the water table depth and its salinity.

Little response of the water table to drip irrigation occurred in these commercial fields, except at one site where over-irrigation occurred part of the time. Although drainage below the root zone occurred under SDI, as indicated by the computer simulations and soil salinity distributions around drip lines (Fig. 10.2), the amount of drainage per irrigation was small because of the small water applications per irrigation and, because of the high irrigation frequency, its distribution over time was relatively uniform. As a result, the natural subsurface drainage in these fields appeared to be sufficient to prevent groundwater intrusion into the root zone. This behavior suggests that, for the conditions found in these fields, subsurface drainage systems and drainage water disposal methods are not needed for properly managed and designed drip irrigation systems. For locations where the water table is affected by drip irrigation, subsurface drainage systems and disposal methods may be required. Similar responses were observed by Ayars et al. (2001). Also, Ayars et al. (1993) provided salt control by leaching with sprinkle or surface irrigation during pre-plant irrigation. There was adequate recession of the water table at the end of the previous irrigation season and pre-plant irrigation to provide storage for deep percolation used for leaching without impacting the following crop.

10.4.1.4 Economic Considerations

The improved irrigation practice must be economical to implement. The best incentive for encouraging subsurface drainage flow reduction through improved irrigation is increased economic benefits, compared to the returns prior to the adoption of a best management practice (BMP). An improvement that reduces economic benefits, even though the benefits are still positive, may be adopted reluctantly, while an improvement that results in a net loss, i.e. cost exceeds revenues, will not be accepted. Improvements that are uneconomical are not sustainable. Public policies may be needed to encourage drainage reduction for conditions where the profit is larger for irrigation methods that generate more drainage water compared to alternative systems. Policy considerations are discussed elsewhere in this book.

Improving furrow irrigation is not likely to improve significantly the economics of furrow irrigation. The improvement will increase capital and labor costs, but it is unlikely that yields will increase significantly, based on improved irrigation studies (Fulton et al. 1991; Styles et al. 1997). A similar situation may exist for converting to sprinkle irrigation from surface irrigation, even though the conversion may increase yields. Wichelns et al. (1996) found that the high labor costs of moving sprinkler lines more than offset any savings, due to reduced water costs when converting from furrow irrigation to sprinkler irrigation.

Converting to drip irrigation has the highest potential of profitability, but the benefits of this improvement cannot be predicted with a reasonable degree of confidence, and thus growers who convert to drip irrigation face some economic uncertainty, especially for lower cash value crops. The field-scale comparisons of drip irrigation and furrow irrigation of cotton showed that uncertainty exists in the economics of drip-irrigated cotton even though cotton yields of drip irrigation were consistently higher and applied water consistently less than for furrow irrigation. However, converting to SDI of processing tomatoes was found to be highly profitable because of the high cash value of the crop and the yield increase under drip irrigation.

An option for improving the economics is to convert to higher cash-value crops with the improved irrigation system. However, higher cash-value crops generally are more salt-sensitive than are the lower cash-value crops, and thus, unless the improved irrigation practice can provide better control of soil salinity, yields may be adversely affected. A limited market may also exist for higher cash-value crops, thus restricting opportunities for converting from low cash value crops to higher cash value ones. Irrigators of lower cash-value crops face a dilemma. Regardless of water costs, water supplies, land quality, adoption of sprinkler and drip irrigation may be uneconomical (Wichelns et al. 1997).

10.4.2 Drainage System Design and Water Table Control

10.4.2.1 Drainage System Design

Subsurface drainage system design for arid and semi-arid irrigated areas has traditionally followed the method developed by the US Bureau of Reclamation (Interior 1993). In this procedure, a cropping pattern is established and the appropriate irrigation and deep percolation schedule are computed. The drainage system design is based on the maximum calculated deep percolation in the crop rotation. This procedure uses the concept of dynamic equilibrium, which means that the water table position returns to the starting point at the end of the irrigation season. It assumes that the depth of the water table at the mid-point between laterals is closest to the soil surface at the end of the irrigation season. The minimum mid-point water table depth has generally been set at 1.9 m (6.2 ft) to minimize salt movement into the crop root zone. The resulting system is managed to operate with continuous flow from the drainage system. The subsurface laterals are installed such that they are parallel to the grade of the soil surface.

However, quality of drainage water has become a significant environmental problem, since this water may contain salt, nitrate, and possibly toxic trace elements, which has led to a reconsideration of criteria (Guitjens et al. 1997) and designs used for subsurface drainage systems in arid irrigated areas. Drainage water quality, now considered a design criterion (Ayars et al. 1997), is accommodated by reducing the lateral depth, which in turn reduces the lateral spacing, and the net effect is to reduce the depth of the flow lines moving to the drains (Ayars et al. 2006b). In areas with deep salt deposits, such as are found in the SJV, the deep flow lines mine salt from deep in the soil profile and create salt loads that are in excess of what is needed to maintain the salt level in the root zone (Jury et al. 2003; Grismer 1990). An alternative to changing the depth and spacing of the drain laterals is to provide control on the drainage system, which also modifies the flow lines to a shallower depth (Ayars et al. 2006b).

Subsurface drainage systems can no longer be designed as an afterthought to correct poor irrigation system design and management. Irrigation and drainage systems must be designed together, as an integrated water management system with specific management objectives. The irrigation system design will include the required management, which provides the detail needed to calculate potential deep percolation losses and the schedule of loss needed to complete the design of the drainage system.

Use of water table control structures requires modification of the lateral installation, such that laterals are installed perpendicular to the surface grade, which provides a reasonably uniform depth to the water table from the soil surface and enables water table control, without creating a waterlogged condition at the tail end of the field (Ayars 2003). Implementing water table control will also allow improved in-situ use of groundwater by crops, which, in turn, will modify the irrigation schedule and the total water requirement for irrigation (Ayars and Hutmacher 1994). Studies in the SJV have demonstrated significant use of groundwater by cotton (Ayars and Schoneman 1986; Wallender et al. 1979). Other studies have demonstrated significant crop water use from shallow groundwater (Ayars et al. 2006a). The crop water use is affected by the groundwater salinity, crop salt tolerance, depth to groundwater and the irrigation frequency. As the irrigation frequency is decreased, crop water use from shallow groundwater increases (Fig. 10.6). Crop water use will increase as the water table gets closer to the root zone and the soil does not become waterlogged. Matching the crop salt tolerance to the groundwater salinity improves *in-situ* water use. These are the factors that support the integrated management of the irrigation and drainage system and the improved design of that system.

10.4.2.2 Drainage Water Reuse

Reuse of saline drainage water has been determined to be an effective method of source control in the SJV (Ayars et al. 1993). It requires the careful selection of crops and the management of salinity within the system. It also requires a new approach to the management of salt in irrigated agriculture. Previously, salt was a commodity that was to be eliminated from the system, without regard to the potential for environmental consequences. Now, salt needs to be managed within the system, and removal must focus on what is needed to maintain production. This approach will require significantly improved water management to be a sustainable system.

Studies have shown that salinity was not the critical component in reusing saline drainage water for supplemental irrigation; accumulation of trace elements such as boron (B) represents a greater threat to the sustainability of the system. Ayars et al. (1993) demonstrated significant accumulation of B at the end of 5 years after using saline water containing 5 mg L^{-1} B for supplemental irrigation, including regular use of pre-plant irrigation to manage the salt during this period. There were no indications of yield loss at the end of this period, but there were indications of plant damage.

Surface discharge of saline drainage water is no longer possible in much of the SJV due to water quality restrictions on salt and Se loading in the San Joaquin River. As a result, many farmers will be required to implement on-farm disposal of drainage water. This will require implementation of source control methods, developing water table management strategies, and reusing saline drainage water. The question remains as to whether terminal evaporation facilities will be required or whether the combination of improved water management, coupled with water table control and reuse, will be sufficient to meet the short-term requirements for disposal. Studies have demonstrated that with the use of SDI, it is possible to grow crops in areas with shallow, saline groundwater that are not managed with a subsurface drainage system. This suggests that the combination of improved water management and lateral and vertical drainage flow may make it possible to sustain irrigation and manage salinity.

10.5 Recommended Practices

10.5.1 Irrigation System Selection and Management

Recommended BMPs for reducing subsurface drainage through improved irrigation:

- Use sprinkle irrigation for pre-plant irrigations and the first crop irrigation if needed.
- Use sprinkle irrigation or furrow irrigation with reduced furrow lengths for crop irrigations.
- Convert to drip irrigation. Recommendations for sustainable SDI of processing tomatoes or other moderately salt-sensitive crops under saline, shallow groundwater conditions include the following cultural practices:

  1. Seasonal water applications should be equal approximately to the seasonal ET. This amount of water provides sufficient localized leaching. Higher applications could raise the water table and cause saline groundwater intrusion into the root zone; smaller applications may decrease yield.
  2. Daily to two to three irrigations per week should occur. Daily irrigations are recommended for very shallow, saline groundwater to minimize the potential upflux of saline water.
  3. The EC_i should be about 1.0 dS m^{-1} or smaller. Higher EC levels may reduce yield of salt-sensitive to moderately sensitive crops.
  4. Periodic leaching of salt accumulated above subsurface drip lines will be necessary with sprinklers for stand establishment, if winter and spring rainfall is insufficient.

- Drip irrigation systems should be designed for a high uniformity of applied water.
- Drip irrigation systems should be properly maintained to prevent emitter clogging.

Use appropriate irrigation scheduling practices and soil moisture monitoring to apply the right amount of irrigation water. The data in Table 10.1 can be used to account for crop water use of the shallow groundwater for some crops.

10.5.2 Drainage System Design and Management

Christen and Ayars (2001) developed a set of BMPs, including a set of guiding principles, for the design, implementation, and management of subsurface drainage systems for use in arid and semi-arid irrigated areas, which are summarized below:

- Drainage should be implemented only after it has been determined that improving irrigation system design and operation will not be adequate to protect the crop and sustain irrigation
- The drainage system design must be site-specific for the crops and soils and not based on regional rules of thumb.
- The system design should provide adequate salinity control for the crop root zone, while minimizing the mobilization of salt below the root zone.
- The drainage system design must include components for monitoring and managing the drainage flow and water table position, such that the drainage system is part of an integrated water management system.
- The drainage system design must provide an environmentally sound method of drainage water disposal. The drainage system owners are responsible for the maintenance and management of the system and the safe management of the water generated by their system.

Additional BMP's are provided for the selection, design and management of subsurface drainage systems. These are detailed below.

10.5.3 Selection BMPs for a Subsurface Drainage System

- Determine the objective for the drainage system
- Assess the limits of the area requiring drainage in the context of the regional hydrogeology and land uses.
- Local and site specific conditions must be used in the design.

10.5.3.1 Design BMPs

1. Develop the drainage design criteria, using minimum leaching fractions, minimum rates of water table drawdown and minimum water table depths.
2. Design should incorporate a temporal analysis of deep percolation from selected irrigation systems for the critical crop(s), assuming BMPs.
3. Design should include water table control components.
4. Design should be based on management units that relate to the irrigation system layout.
5. Design must include effluent management and monitoring plans, based on identified salt, nutrient, and other constituents of concern in the drainage water.
6. Systems designed for reclamation should include provisions for future management.

10.5.3.2 Management BMPs

1. Discharge should be managed to prevent flow, other than what is needed to maintain desired water table position or leaching fraction.
2. Existing drainage systems should be retrofitted, when possible, to include management structures and be formed into management units.
3. System should be controlled to promote in-situ crop water use from shallow groundwater, when feasible.

Examples of application of the BMPs and additional discussion of these BMP's are available in Christen and Ayars (2001).

10.5.4 Drainage Water Reuse

Drainage water reuse will become an integral part of the farm water management strategy for disposal of saline drainage water. Reuse of saline drainage water should employ surface irrigation methods, since this will aid in maintaining leaching fractions and in salinity control. Several options for reusing drainage water are discussed in Rhoades et al. (1989). The approach used in these studies was to alternate good quality water for pre-plant irrigation followed by undiluted saline water for irrigation after stand establishment. Others studies have evaluated the blending of drainage water to reduce salinity or the cyclic use of saline drainage water and low salinity water, depending on the crop. Given the high levels of soil salinity on the west side of the SJV, the use of good water for pre-plant leaching and stand establishment followed by direct use of saline will be the best method.

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Chapter 11
Drainage Water Reuse: Concepts, Practices and Potential Crops

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11.1 Introduction

Reuse of drainage water for irrigation is recognized as a viable means of reducing the amount of saline-sodic spent water that will ultimately require treatment or disposal in the western San Joaquin Valley (SJVDP 1990). This practice is not a long-term solution in itself, but rather an integral component to drainage water management in the San Joaquin Valley. For reuse to be successful, soil salinity and boron (B) cannot accumulate to levels damaging to crop growth; soil physical conditions conducive to water infiltration must be sustained; and trace element accumulation in crops and forages must remain low enough not to threaten the health of humans or livestock (Oster and Grattan 2002).

Use of saline or saline-sodic water for irrigation requires special consideration in management practices, such as selection of appropriate crops and crop rotations, improvements in water and soil management, and in some cases, the adoption of advanced irrigation technology. Management must achieve salinity control within the root zone, avoid deterioration of soil physical conditions, and minimize the accumulation of certain trace elements (e.g. B, selenium [Se], and molybdenum [Mo]).

The University of California's Salinity Drainage Research Program recognized the role of drainage water in the overall management plan. Therefore, many studies were funded by the Task Force to improve understanding and demonstrate the

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_11,

feasibility of drainage water reuse and management in the San Joaquin Valley, as well as to identify potentially new crops in reuse systems. Research was conducted in the field, in the greenhouse, and by computer simulations. This chapter provides a brief overview of the highlights and accomplishments of these research efforts.

11.2 Drainage Reuse Strategies

Several methods of utilizing saline water have been tested experimentally or demonstrated under field conditions. The methods differ regarding where, when, or how the saline water is applied and whether non-saline water is available or substantial rainfall occurs during the year. A short description of each is provided below, but more detailed information is found in reviews (Grattan and Oster 2003; Oster and Grattan 2002).

11.2.1 Blending Water Supplies

One common way of improving the quality of saline water for irrigation is to blend it with water of lower salinity. The goal is to produce an irrigation water of suitable quality by mixing two source waters, while, at the same time, expanding the overall irrigation water supply. The suitability of the blended water depends on the salt tolerance of the crop being irrigated. Shalhevet (1984) discussed two blending processes: network dilution and soil dilution. With network dilution, water supplies are blended in the irrigation conveyance system to achieve the targeted blended ratio. With soil dilution, the soil acts as the medium for mixing water of different qualities. Waters of different qualities are alternated, according to availability, between irrigations or within an irrigation event. Dinar et al. (1986) described a method to calculate optimal ratios of saline and non-saline irrigation water for crop production.

Blending saline drainage water with good quality non saline water is neither always economically feasible (Dinar et al. 1986) nor does it unconditionally increase the usable water supply (Grattan and Oster 2003). There is an upper salinity threshold for water suitable in blending. Higher quality waters that are blended with a saline water, resulting in a blend whose quality exceeds a crop's salinity tolerance, will actually reduce the overall 'available water' supply. Intuitively, blending is not an attractive alternative if the saline water does not make up at least 25 % of the total irrigation water requirement. The risks of potential crop damage and additional costs incurred in management, associated with a more saline irrigation supply, would likely outweigh the benefits from a modest increase in available water supply.

11.2.2 Cyclic Use of Saline and Non-saline Water

With a cyclic strategy, the soil salinity is first reduced purposefully by irrigation with good quality water, thereby facilitating germination and permitting crops with lesser tolerances to be included in the rotation (Rhoades et al. 1992). At the early stages of plant growth, the goal is to keep the average salinity low in the upper portion of the root zone, a region critical for emergence and plant establishment. Once established, plants are less susceptible to damage from salinity.

The cyclic strategy was introduced and tested by Rhoades (1984). In this method, for selected crops, saline water is used during certain portions of their growing season, while non-saline water is used at other times. The objective is to minimize soil salinity (i.e. salt stress) during the salt-sensitive growth stages, or when salt-sensitive crops are grown in a rotation of crops. This does not necessarily imply that saline drainage water is only applied to salt-tolerant crops after they reach a salt-tolerant growth stage. Soil salination lags behind saline water application, allowing a more salt-sensitive crop to be irrigated with saline water in a soil that is initially non-saline (Shennan et al. 1995; Bradford and Letey 1992). Likewise, without pre-plant leaching by irrigation or rainfall, it may be difficult to return quickly to a salt-sensitive crop in the rotation, once the soil profile is salinized.

The yield-threshold levels of salinity are determined from controlled studies and reflect the response of crops to the average root zone salinity after the establishment of seedlings using non-saline water. However, the salt tolerance of many crops increases as the plant matures (Maas and Grattan 1999). For this reason, crops can tolerate relatively higher salinity levels late in the growing season without suffering a loss in yield. Furthermore, applying saline water later in the season reduces crop exposure time to salinity, allowing waters of higher salinity to be used.

11.2.3 Sequential Use

In sequential reuse systems (Fig. 11.1), salt-sensitive crops are irrigated with non-saline water. The drainage water collected from these fields is then used to irrigate more salt-tolerant crops on other fields. This reuse system may be designed at the farm or regional scale and may have one or more sequential fields that reuse drainage water. Unlike the cyclic strategy, a given land area is dedicated to either salt-sensitive or salt-tolerant crops.

The main purposes of sequential reuse are the following:

- Reduce the soil salinity in fields that are tile drained and irrigated with non-saline water, thereby increasing their productivity,
- Obtain an economic benefit by using drainage water for crop production,
- Increase the area planted to high value salt-sensitive crops and reduce the volume of drainage water required disposal.

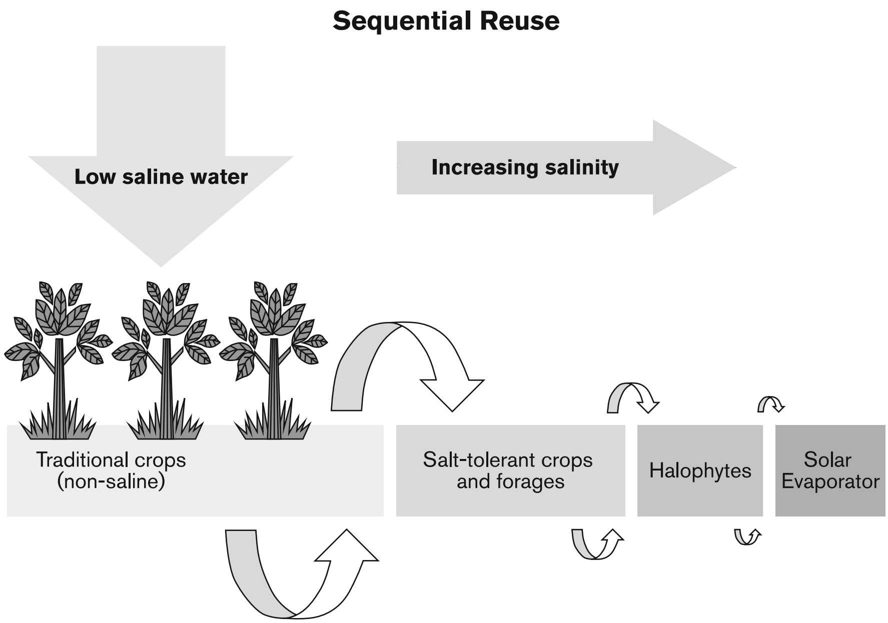

Fig. 11.1 Schematic representation of a sequential drainage water reuse system

Such sequential reuse systems have been tested and are implemented in the San Joaquin Valley. The studies include multiple options for managing salinity, including a wide range of plant species in the rotation having various tolerances to salinity, including forages and halophytes. These studies also explore the potential uses of salt harvested from the terminal solar evaporator (Ayars and Basinal 2005).

The sequential reuse system can be beneficial to farms that are not allowed to discharge subsurface drainage water. It allows a drainage system to be installed for lowering the saline water table and leaching salts from the area irrigated with non-saline water. As such, this increases the productivity on this portion of the farm. The concept is to increase the salinity sequentially and to reduce the volume of drainage water that will ultimately need safe disposal.

Although sequential reuse is conceptually attractive, there is a substantial lag time for salts at the beginning of the reuse sequence to reach the final stage. Jury et al. (2003) conducted a study using a transfer function model, assuming typical drain-line spacing and water management practices, and found that such a reuse system would never effectively reach steady-state, but rather it could take decades or much longer for water and dissolved salts to move though the sequential system. Because low-lying, fine-textured soils are often the site where such systems are being considered, travel times may be even longer. Therefore, caution is advised for those designing sequential reuse systems and estimating the rate of salt movement through the sequential system, particularly if steady-state assumptions are used. Such an assumption will result in poorly designed sequential reuse systems. Drainage water reuse systems are subjected to fluctuating water tables, due to off-farm conditions. Water tables are affected by regional groundwater extraction and by excessive leaching. These fluctuations, particularly where the water table depth is below the tile lines, will also affect the time needed to establish quasi steady-state conditions.

A regionalized approach has been adopted by others, in which salt-tolerant forages (e.g. Bermuda grass, *Cynodon spp*.) are grown on salinized land irrigated with water of variable quality, including saline drainage (Corwin et al. 2003, 2006; Kaffka et al. 2002, 2004). These ongoing studies have shown that beef cattle can graze successfully on Bermuda grass as a sole source of feed during much of the year.

Drainage water reuse strategies are not mutually exclusive; a combination may be most practical in some cases. For example, within a sequential reuse scheme, blending and/or cyclic methods may be used occasionally for salinity management, depending upon water availability, or depending on the objective, such as to germinate and establish crops.

11.3 Drainage Water Reuse Studies: Field and Computer Simulations

11.3.1 Short and Long-Term Field Studies

In the past 20 years, a number of field studies have been conducted that tested various management strategies for irrigating crops with saline or saline-sodic water. This section highlights key findings from these studies.

The earliest field studies on drainage water reuse pre-dated UC Salinity Drainage Research Program. Rhoades (1984, 1987) and Rhoades et al. (1988, 1989) tested the cyclic strategy of using waters of different salinities and found it to be sustainable in maintaining crop rotations that included both moderately salt-sensitive and salt-tolerant crops. The non-saline water was used for pre-plant and early crop irrigations of the moderately salt-tolerant crops and for all irrigations of the moderately salt-sensitive crops. Salt-tolerant crops were irrigated with saline water after they were well established. After the salt-tolerant crop was grown, a pre-irrigation with low-salinity water reclaimed the upper portion of the soil profile in order to establish the salt-sensitive crop.

Rhoades (1984, 1987), conducted cyclic reuse studies in California's Imperial and San Joaquin Valleys (SJV). In a successful test of the cyclic strategy conducted in the SJV of California, non-saline water (EC $= 0.5$ dS m^{-1}) was used to irrigate cotton during germination and seedling establishment; water with EC $= 7.9$ dS m^{-1} and SAR (sodium absorption ratio) $= 11$ was used thereafter (Rhoades 1987). Wheat was subsequently irrigated with the 0.5 dS m^{-1} water, followed by 2 years of sugar beets with the cyclic strategy used again for irrigation. Rhoades et al. (1988, 1989) reported the results from a second study conducted in the Imperial Valley of California. In a rotation of wheat, sugar beets, and melons, Colorado River water

(EC = 1.5 dS m^{-1}, SAR = 4.9) was used to irrigate cantaloupe, a moderately salt-sensitive crop, and for the pre-plant and early irrigations of wheat and sugar beets. Alamo River drainage water (EC = 4.6 dS m^{-1}, SAR = 9.9) was used for all other irrigations. Sugar beet and wheat yields were not reduced, and crop qualities were often improved by the use of saline drainage water. These investigators found that saline water could be used successfully for irrigation of crops, including a rotation of salt-tolerant crops and moderately salt-sensitive crops.

Ayars et al. (1990, 1993) used drip irrigation for three consecutive years to apply a 7–8 dS m^{-1}, SAR = 9 water to cotton, after it was established with a 0.4–0.5 dS m^{-1} water. The saline water supplied 50–59 % of the irrigation water requirement. A wheat crop irrigated with the 0.5 dS m^{-1} water followed cotton; sugar beets followed wheat and were irrigated with the 8.0 dS m^{-1} water after stand establishment. Yields under these conditions were the same as from continuous irrigation with good-quality water. The investigators did note, however, a gradual increase in soil B concentrations over time from drainage water application (5–7 mg L^{-1} B), despite annual rainfall and preseason applications of non-saline water in excess of 150 mm.

Others have expanded on the cyclic reuse approach. Shennan et al. (1995) tested two cyclic drainage-water reuse practices on processing tomato in a 3-year rotation with cotton over a 6-year period. In both practices, drainage water was applied to processing tomato after first flower to take advantage of salinity's enhancement of fruit quality and continued to the end of the season. In one practice, non-saline water was used at all other times (i.e. saline water applied 1 out-of-3 years). In the other practice, drainage water (EC = 7.4 dS m^{-1}, SAR = 12) was also applied to the following cotton crop, after thinning, while non-saline water was applied at all other times (i.e. saline water applied 2 out-of-3 years). Non-saline water (EC = 0.4 dS m^{-1}, SAR = 1.6) was used as the source of irrigation water at other times, and both reuse practices were compared to rotations in which only non-saline water was used. In the practice where drainage water was applied 1-out-of-3 years, yields of tomatoes were sustained, even though drainage water supplied up to nearly 60 % of the irrigation water requirements. In the cyclic reuse treatment where drainage water was applied 2-out-of-3 years, tomato yields were reduced in 2 of the 6 years. Soluble solids in tomato fruit, on the other hand, were increased 5 of the 6 years in plots where drainage water treatments were applied.

Similar results were found with tomatoes in field studies conducted in different locations in the SJV but for only 1 year when drainage water supplied more than 65 % of the irrigation water requirement (Grattan et al. 1987). Pasternak et al. (1986) also reported an increase in soluble solids in tomatoes when irrigated with an EC = 7.5 dS m^{-1} water, after the fourth or eleventh leaf stage, but yields were reduced by 30 %. Differences between these studies may be due, in part, to differences in the anion composition of the saline water. In Israel, where Pasternak and colleagues conducted their work, the saline water is normally chlorine-dominated; whereas, in the SJV of California, the saline drainage water is sulfate-dominated.

The study by Shennan et al. (1995) also examined the behavior of salts and B over time at different depth increments. On a relative basis, salts were more readily leached than B. At the 60–140 cm depth interval, the ECe (electrical conductivity of the soil saturation extract) in plots irrigated with saline water increased the first year

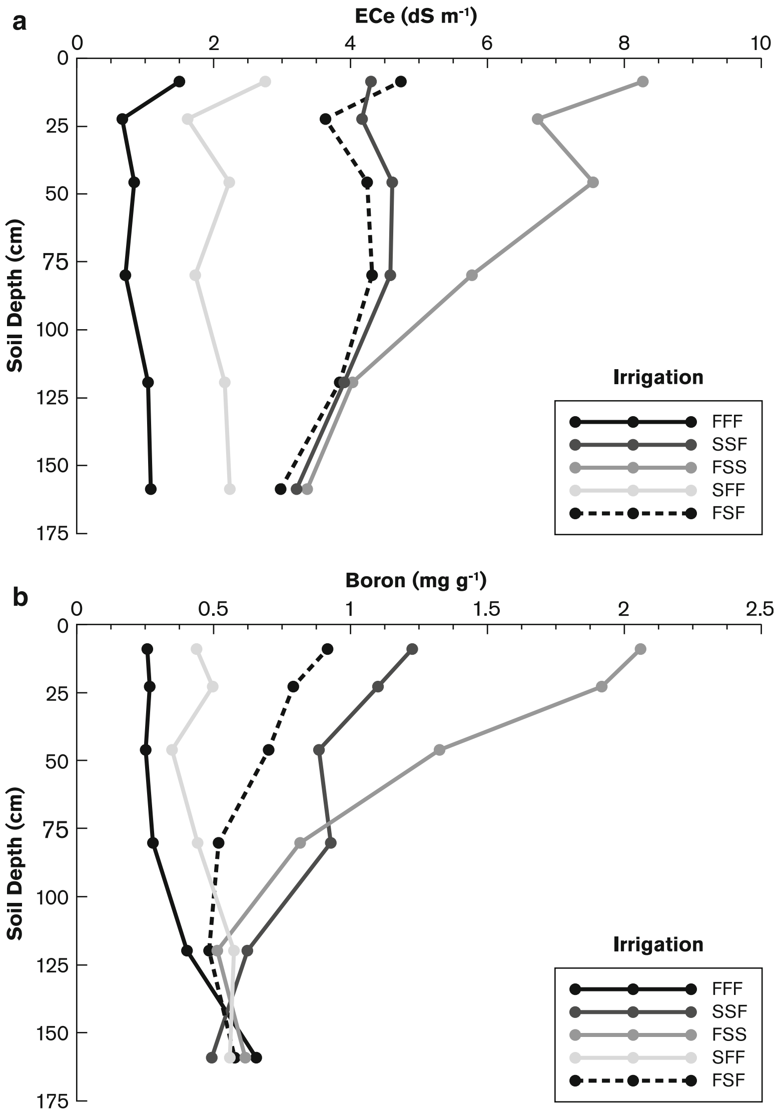

Fig. 11.2 Electrical conductivity, ECe, and boron concentration of soil saturated extract in cotton plots subjected to 3-year cyclic reuse rotations. Legends *F* and *S* represent non-saline 'fresh' water and saline water application, respectively in annual sequence (Adapted from data in Shennan et al. (1995))

(1986 data not shown). However, the concentrations of B at this depth were not found to increase until 1988 in plots that received drainage water 2-out-of-3 years and were not found to increase until 1989 in plots that received drainage water 1-out-of-3 years. In the upper 15 cm of the soil profile, the ECe increased after saline water irrigation, but then, after 2 years of irrigation with non-saline water, the ECe returned to the level found in the control. Figure 11.2a, b illustrate the distribution of salinity and B, respectively, after 3 years of cyclic reuse in these long-term plots. The level of salinity or B in plots was related both to the fraction of the drainage water applied, as well as the timing of applications. For example, plots that received saline drainage water in 2 of the last 3 years (FSS denotes fresh water [F] in year 1, followed by saline drainage water [S] in years 2 and 3) had nearly twice the salinity as those that received drainage water the first 2 out of 3 years (SSF) in the upper 50 cm. Furthermore, when fresh water was applied after saline water, the salinity in the plots was reduced substantially (e.g. FSS v SSF and FSF v SFF). Similar patterns were observed for soil B concentration, although most of the changes occurred only in the top meter of the profile, as compared to changes in the entire 160 cm profile for salinity.

As indicated earlier, the feasibility of cyclic reuse practices depends, in part, on changes in soil chemical and physical properties over the long runs. In a long-term cyclic study conducted on a clay loam soil by Shennan et al. (1995), no difference in water infiltration rates was found, measured with a steady-state infiltrometer, between plots that received saline water and those that did not. Nevertheless, they did find a significant reduction in cotton stands in 1988 in plots that were salinized the previous year.

Use of saline drainage water (TDS = 9,000 mg L^{-1} and SAR = 16–30) on a clay soil also caused a reduction in stands in fields planted to cotton and safflower (Rains et al. 1987) within 2 years. The inability to prepare a seedbed with the tilth necessary for adequate seed emergence contributed to poor cotton stand establishment (Oster 1994). Mitchell et al. (2000) found a reduction in cotton seedling establishment related to irrigation with saline-sodic water. Surprisingly, the inclusion of cover crops to improve soil physical conditions actually aggravated the situation. These investigators suggested stand establishment was reduced by the formation of “stubble-reinforced” surface crusts.

Kaffka and Hembree (2004) studied the emergence of sugar beet seedlings in salt-affected soils in the San Joaquin and Imperial Valleys (SJV and IV). Different seed treatments were evaluated for their effects on emergence. At the IV site, salinity accumulated at and near the soil surface, due to surface irrigation and upward movement of salt from a shallow, saline water table. Primed seeds treated with imidicloprid, a neonicotinid insecticide, emerged at significantly higher rates at this location than the other treatments used, resulting in superior emergence under transient saline conditions. At the SJV site, salinity in the upper 30 cm of soil ranged from 2 to 10 dS m^{-1} due to prior saline irrigation applications (Shennan et al. 1995; Kaffka et al. 1999; Bassil and Kaffka 2002a, b). Therefore, seedling dry matter accumulation and rates of emergence declined at ECe levels greater than 6 dS m^{-1}. However, final plant populations were not affected over the salinity range. The rate of emergence of primed seeds in saline plots equaled that of non-primed seeds in plots without low levels of salinity.

Kaffka et al. (1999) conducted a study with sugar beets (*Beta vulgaris*) in the same long-term reuse plots used by Shennan et al. (1995). Results were consistent with most of the reports already cited. Transient salinity levels in saline irrigated plots exceeded threshold levels for yield reduction. The ability of this deep-rooted crop to recover water from a larger volume of soil allowed the beets to adapt without adverse effects on root yield. Alternating saline with non-saline irrigation water (using saline water either before or after non-saline in midseason) did not affect root yields. However, due to high concentrations of nitrate (NO_3^-) in the saline well water, late-season irrigation with saline water reduced sugar yields more than early-season irrigation.

High concentrations of nitrate in saline drainage water may be a nutrient resource for crops or may negatively affect crop quality. Nitrogen (N) is often a growth-limiting nutrient so N-fertilizer applications can be adjusted downwards to take advantage of the N applied in drainage water. At the same time the crop is absorbing nitrate from the drainage water, it is also a potential pollutant when drainage water is ultimately treated or disposed. However, too much N in the soil can affect crop quality. For example, Shennan et al. (1995) found that the high NO_3^- concentration in drainage water (1.1 mmol L^{-1}) prolonged reproductive growth in processing tomato, such that the percentage of 'green' fruits was double those from non-saline control plots at the time of harvest. Because sugar beet is a deep-rooted, salt-tolerant crop, it can recover residual soil NO_3^-. Additional NO_3^- in drainage water applied to the crop reduces root sugar content (Kaffka et al. 1999). Therefore, accounting for N and making sure soil levels are sufficiently low near the end of the season, are necessary steps to assure sugar beet root quality. In general, accounting for NO_3^- and other nutrients in drainage water is essential for a prudent drainage water management program.

Following sugar beets, safflower was grown in the same long-term drainage reuse plots (Bassil and Kaffka 2002a, b). Safflower produced less shoot biomass, used less water, and matured earlier than plants irrigated with non-saline water. However, seed yields and overall oil yield were similar in plants sampled from saline and non-saline plots because safflower was able to adjust its harvest index under salt stress.

These short- and long-term studies, many funded by the UC Salinity-Drainage Task Force, indicate that saline-sodic water can be used successfully for irrigation of many conventional crops. Success depends upon (a) the salinity of the drainage water, (b) the proportion that is used relative to non-saline water and rainfall, (c) the crop salt tolerance and (d) when and to what extent the crop is being stressed by excess salinity. By careful management, soil salinity and B can be controlled, and soil physical conditions can be maintained by adequate and timely applications of gypsum.

11.3.2 Computer Simulations

Crop tolerance to salinity, the salinity of the irrigation water, and the amount of water applied are variables that affect achievable yield. Field experiments, which investigate all of these variables, are difficult and expensive to conduct. Therefore, models become a valuable tool because they can be programmed readily for computers to simulate the consequences of different management options.

Models can be broadly classified into two categories: steady-state and transient-state analyses. The use of crop salt-tolerance coefficients, as reported by Maas and Hoffman (1977), is common to all models. All models compute yield relative to the yield that would be achieved by a non-stressed crop, rather than the actual yield. Also incorporated into the models are (a) water balance in which the sum of irrigation and precipitation must equal the sum of evapotranspiration, (b) deep percolation below the root zone, and (c) change in water storage in the root zone. However, the steady-state analysis inherently assumes constant soil-water and soil-salinity status so that the change in water storage is zero.

Several factors that may differ between models should be considered in evaluating a given model:

- The fact that evapotranspiration is a variable associated with plant growth as well as climatic conditions is ignored in some models and leads to errors. Plants have a basic protective mechanism. When soil salinity reaches a level that decreases plant growth, the plant transpires less water. Reduction in transpiration causes an increase in deep percolation, allowing more salt to be leached from the root zone. The net result reduces the soil salinity to which the plant is exposed.
- Soil salinity is almost never constant throughout the root zone. Usually, salinity increases with depth in the root zone. Some models assume that plants respond to the linear average root zone salinity. Others assume that the response is to the weighted water uptake distribution in the root zone. The latter assumption leads to higher predicted yields than the former. Neither, however, accounts for the fact that plants extract extra water from non-stressed root zone areas to compensate for reduced water uptake from the stressed zone of the roots.
- Rainfall, during the crop and non-crop season, diminishes the impact of salinity and must be accounted for in evaluating the impact of irrigating with saline waters.
- Because all models assume uniform water application across the field, the effects of the irrigation uniformity assumption must be included in the final analysis. However, actual water distribution, not a coefficient, is required for an accurate assessment.
- The factors listed above should be considered in evaluating any model. We will discuss these in the context of two models, which were developed as part of the University of California Salinity-Drainage Program. Letey et al. (1985) developed a steady-state model that accounts for the reduction in transpiration as plants become stressed by soil salinity. The model assumes that crops respond to the linear average root zone salinity. Letey and Dinar (1986) reported the relationship between the relative yield of several crops and the amount of applied water of a given salinity. The assumption that the plant responds to the linear average salinity leads to an over-prediction in the amount of irrigation required to achieve maximum yield. However, because the model does consider

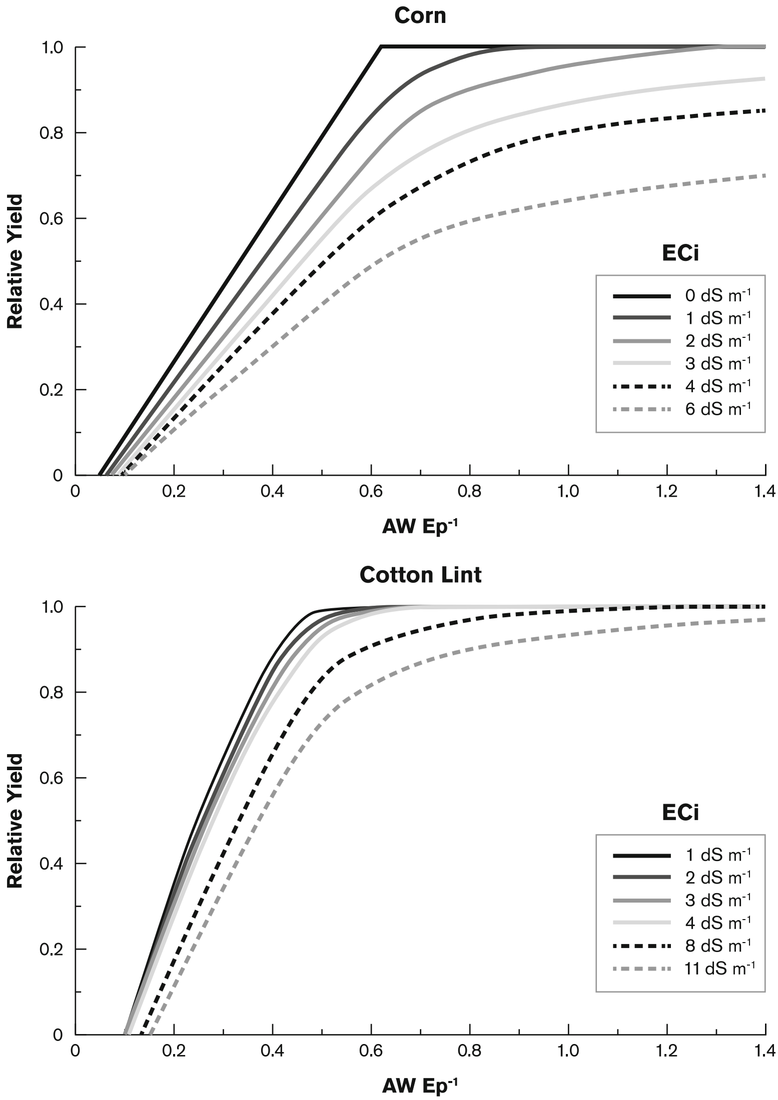

Fig. 11.3 Relative yields vs. salinity of irrigation water for corn (ECi = 1–6 dS m^{-1}) and cotton (ECi = 1–11 dS m^{-1})

reduced transpiration after a crop is stressed, the model leads to more accurate yield predictions that are less than the maximal. The report was useful for comparing several crops for which simulations were conducted. For example, Fig. 11.3 illustrates that cotton can be irrigated with fairly high saline water with very little reduction in yield, and maximum yield of cotton can be achieved with very small increase in the amount of irrigation. Conversely, corn is impacted to a far greater extent than cotton when irrigated with saline water.

The effect of rainfall cannot be determined directly from a steady-state model. However, by calculating the volume-weighted average of the irrigation water's salinity and rainfall (zero salinity), the effect of rainfall can be approximated. This salinity would be used, rather than the irrigation water salinity, for interpreting the graphs, which are depicted by Letey and Dinar (1986). For example, assume that the irrigation water salinity was 2 dS m^{-1} and the annual rainfall was equal to the amount of irrigation, then the salinity of 1 dS m^{-1} would be used.

Cardon and Letey (1992) reported a transient-state model that accounted for the reduced transpiration associated with a stressed crop. Their model was refined, and the opportunity for the crop to remove more water from the non-stressed portion of the root zone to compensate for the stressed portion of the root zone was reported by Pang and Letey (1998). Good agreement was found between the simulated corn yield from the model and experimentally measured yield in an experiment where both salinity and water stress were imposed on the crop (Feng et al. 2003).

A recent comparison between results achieved from steady-state and transient-state models was done by Letey and Feng (2007). The steady-state models generally over-predicted the negative consequences of irrigating with saline waters, as compared to the transient-state models. Guidelines that have been developed for irrigating with saline waters and adopted by many are based on prior steady-state analyses (Ayers and Westcot 1985), and may be too conservative. The recent research finding suggests that reuse of drainage water for irrigation may have greater opportunity than previously expected.

11.4 Potential Problems Associated with Trace Elements

11.4.1 Accumulation of Selenium and Molybdenum in Crops

Naturally occurring trace elements in soils and in the underlying, shallow groundwater add another dimension to the management of saline drainage waters (van Schilfgaarde 1990). Trace elements, particularly selenium (Se) and molybdenum (Mo), occur in the western SJV of California and pose a potential threat to the sustainability of irrigated agriculture in this area. Selenium and Mo are important since they are found in relatively high concentrations at many locations in geochemically mobile forms (i.e. SeO_4^{2-}[selenate] and MoO_4^{2-}[molybdate], respectively). These chemical species are also the forms in which they are biologically available.

Plant uptake of Se and Mo is the primary process by which these essential trace elements for humans and animals are introduced naturally into the diet from the terrestrial environment. However, a number of factors affect trace element absorption and accumulation in plants. Although it is known that plants vary widely in their ability to accumulate these trace elements, physical and chemical factors, such as pH, the presence or absence of sulfate, soil texture, and organic matter all can influence uptake as well (Mikkelsen et al. 1989).

Table 11.1 Selenium concentration (in mg kg^{-1} dry weight) in the edible portion of vegetable crop when irrigated with either non-saline California aqueduct water (ECw = 0.3 dS m^{-1}, Se = <2 μg L^{-1}, SO_4^{2-} = 15 mg L^{-1}) or saline drainage water (ECw = 7.1 dS m^{-1}; Se = 45 μg L^{-1}; SO_4^{2-} = 1,000 mg L^{-1}) that were spiked with Se (as selenate) injected into the drip lines Burau et al. Unpublished data

	Se concentration of non-saline aqueduct water (μg L^{-1})			Se concentration of saline drainage water (μg L^{-1})		
Vegetable[a]	0	30	150	45	165	600
Bean, pink	17 ± 7	951 ± 149	2,313 ± 292	100 ± 24	437 ± 63	585 ± 155
Bean, snap	18 ± 3	1,364 ± 267	2,508 ± 453	81 ± 19	466 ± 126	696 ± 111
Cantaloupe	10 ± 4	610 ± 122	2,483 ± 1,081	62 ± 17	435 ± 44	532 ± 110
Corn	27 ± 21	647 ± 128	1,691 ± 189	69 ± 12	465 ± 80	520 ± 33
Cucumber, small	12 ± 4	1,192 ± 41	2,525 ± 375	110 ± 36	614 ± 68	1,017 ± 267
Cucumber, large	7 ± 3	1,543 ± 257	2,878 ± 307	168 ± 39	703 ± 50	931 ± 222
Eggplant	18 ± 10	353 ± 94	2366 ± 911	52 ± 9	352 ± 74	701 ± 104
Pepper, green	10 ± 6	1,024 ± 218	5,136 ± 1555	79 ± 22	603 ± 37	1,037 ± 113
Potato	10 ± 7	447 ± 67	1746 ± 207	168 ± 66	492 ± 87	499 ± 119
Tomato, fresh	35 ± 19	626 ± 70	2,132 ± 374	211 ± 260	628 ± 92	865 ± 138
Tomato, process	40 ± 20	604 ± 110	1,813 ± 345	107 ± 14	626 ± 91	915 ± 85
Zucchini, small	63 ± 21	1,190 ± 599	2,130 ± 280	146 ± 43	864 ± 129	1,187 ± 134
Zucchini, large	26 ± 11	1,072 ± 244	2,564 ± 328	90 ± 25	836 ± 140	1,070 ± 136

[a]Selenium concentration of harvested vegetables are expressed in mean ± standard deviation (mg kg^{-1} dry weight)

Selenium concentrations in plants vary widely among plant types, plant organs and location. In a survey of more than 400 crops, Se content in the edible portion of crops varies widely (<50 to >1,000 μg kg^{-1} dry weight) but no crop contained potentially health-hazardous levels of Se. A number of greenhouse and field studies have been conducted that have measured the accumulation of Se in edible tissues of crops (Tanji et al. 1988; Valoppi and Tanji 1988). Field studies included experiments in which crops where grown in high-Se soils (Bañuelos et al. 2003) and/or where crops were irrigated with saline drainage water containing high levels of Se. The general trend was that the Se concentrations were higher in the leaves and stems than in the fruit or grain.

Plant samples were also collected from various tree crops, including almond, pecan, pistachio and walnut, grown in Western Fresno and Merced County. These investigators found that Se concentrations were highest in leaf tissue, lower in kernels, and least abundant in the shells.

A comprehensive field study was conducted by the Salinity-Drainage Task Force to evaluate dietary concerns related to Se accumulation in vegetable crops (Table 11.1). The results indicated that the presence of sulfate (SO_4^{2-}) dramatically reduced SeO_4^{2-}uptake, resulting in lower concentrations of Se in plant tissues. For example, the Se concentration in vegetable crops was 2.5–8.5 times lower in reproductive tissue in plants irrigated with 165 μg Se L^{-1} (in the presence of high sulfate, 1,000 mg $SO_4^{2-}L^{-1}$), compared to those irrigated with 150 μg Se L^{-1} under non-saline conditions with low sulfate (15 mg $SO_4^{2-}L^{-1}$).

The Se concentration range in edible portions of plants in field experiments and surveys reviewed by Valoppi and Tanji (1988) was 0.002 to <5.0 mg Se kg$-^{1}$ dry weight. A health assessment indicated that crops consumed at expected levels contribute minor amounts to the daily Se intake of humans and do not pose a health risk.

As important as dietary concerns for humans are, potential toxicological problems in livestock, whose diets may rely almost entirely on forage grown in high Se and Mo areas within the SJV, are also critical issues. Selenium and Mo are essential elements for the diets of animals, and the concentration range in forage is rather narrow between deficiency and toxicity to livestock (James et al. 1968; Ohlendorf 1989; Osweiler et al. 1985). Selenium toxicity, such as alkali disease, can occur in livestock that graze on forage containing high levels of Se (Rosenfeld and Beath 1964). The National Research Council (NRC) set the maximum tolerable concentration (MTC) for Se in beef cattle at 2 mg Se kg^{-1} DW (NRC 1996), but for dairy cattle, 5–40 mg Se kg^{-1} was listed as causing chronic toxicity (alkali disease) in several weeks to months (NRC 2001).

In the SJV, important work was carried out to evaluate the effects of high soil Se concentrations on grazing cattle. Cattle grazed on forage grown in the former "Pond 12" at the Kesterson Wildlife Refuge for approximately a 6-month period in 1999. During this period, forage Se concentrations exceeded 1 mg Se kg^{-1} DW. Blood and fecal Se concentrations were elevated but none of the cattle tested exhibited any signs of Se toxicity; their general health was not adversely affected. Additional research must be continued in this area, as well as applying research experiences from around the world to the conditions in the SJV (Sharma and Tyagi 2004; Qadir and Oster 2004).

Molybdenosis is a nutritional disorder that ruminant animals, particularly sheep and cattle, may develop if the animals feed on forage that contains high levels of Mo (Kubota and Allaway 1972; Ward 1978). Molybdenosis results from a Mo-induced copper (Cu) deficiency and is often called a molybdenum-induced hypocuprosis (Mason 1990). It can also delay first estrus in cattle and reduce the pregnancy rate when Mo concentrations in forage are as low as 5 mg Mo L^{-1} (Phillippo et al. 1987). Although Cu supplements can treat deficiency symptoms easily, fertility symptoms appear to be related directly to the Mo concentration in the forage and are not readily correctable.

11.4.2 Potential Limitation to Boron

Boron (B) is essential for cell wall structure and plays an important role in membrane processes and metabolic pathways (Läuchli 2002; Brown et al. 2002). However, there is a small range in which concentrations in soil solution are optimal (Gupta et al. 1985). Above this range, B becomes toxic and below it, B is deficient.

Toxicity can occur in crops when B concentrations increase in young developing tissue or when the margins of mature leaves have lethal levels, but plant-tissue analyses can only be used as general guidelines for assessing the risk of B toxicity (Nable et al. 1997).

Under some circumstances, B may be a concern where saline-sodic water contains high amounts of this potentially toxic element. Such is the case in shallow groundwater on the west side of the SJV in California where B concentrations typically range from 2 to 10 mg L^{-1} (SJVDP 1990). Despite these levels of B in the drainage water in the SJV, B toxicity has not been reported yet on annual crops. In the SJV, saline-sodic drainage water containing 7–10 mg B L^{-1} has been used to irrigate cotton, melon, sugar beet, tomato, safflower, and wheat (Ayars et al. 1990, 1993; Bassil and Kaffka 2002a, b; Grattan et al. 1987; Kaffka et al. 1999; Mitchell et al. 2000; Rhoades et al. 1988; Shennan et al. 1995). At least part of the success may be attributed to the fact that cotton, sugar beet, and tomato are tolerant of B (Maas and Grattan 1999). An additional factor is that rainfall reduces the B hazard, a factor normally not taken into account when assessing B hazards. Moreover, B toxicity is most commonly diagnosed by visible leaf injury. Leaf injury and yield are not invariably related, in part because leaf injury usually occurs late in the growing cycle. Also, interactions between salinity and B on crop growth and yield are not well understood (Lauchli and Grattan 2007); some crops are able to remobilize B within the plant, such that younger tissue will have higher B concentrations than older tissue (Brown and Shelp 1997). In these instances, B injury is expressed as injury to meristematic tissues (developing leaves and buds). Others have described such injury in deciduous trees as 'twig dieback' (Ayers and Westcot 1985).

In salinity-B studies carried out in sand tanks at the US Salinity Laboratory (Riverside, California), investigators found that increased salinity, regardless of its composition, reduced B accumulation in broccoli and its toxic effect. Additional studies were conducted on cucumber, as well as broccoli, under conditions of variable pH. They found that pH has a profound influence on this relationship. For example, as pH increased from slightly acidic (pH 6.5) to slightly alkaline (8.0), a pH typical of SJV drainage waters and soil solutions, cucumber performance was worse overall, but increases in B were much more damaging at the lower pH than at the higher pH (Läuchli et al. 2008). For broccoli, high B concentrations (21 mg L^{-1}) were detrimental only at high pH and only in low salinity (EC = 2 dS m^{-1}) or high salinity (EC = 14 dS m^{-1}) conditions. Increased B was not detrimental at either pH at intermediate salinity levels. It is likely that different mechanisms are operating to affect plant growth, reproductive yield, and internal ion relations (Läuchli et al. 2008).

11.5 Search for Appropriate Crops in Reuse Systems

A number of conventional crops currently grown in the SJV (e.g. cotton, melon, safflower, sugar beet, tomato, and wheat) were tested and found to be successful in both short- and long-term drainage water reuse studies (Ayars et al. 1990, 1993;

Table 11.2 Criteria for selecting crops for saline water reuse

Selection criterion	Desirable characteristics	Undesirable characteristics
Economic value/ marketability	High value/marketable	Low value/unmarketable
Crop salt tolerance	Salt tolerant	Salt sensitive
Biomass production	High biomass production	Low biomass production
Crop boron/chloride tolerance	Chloride and boron tolerant	Chloride and boron sensitive
Crop potential to accumulate toxic element	Excludes toxic element	Accumulates toxic element
Crop quality	Improved or unaffected by saline water irrigation	Adversely affected by saline water irrigation

Bassil and Kaffka 2002a, b; Grattan et al. 1987; Kaffka et al. 1999; Mitchell et al. 2000; Rhoades et al. 1988, 1989; Shennan et al. 1995). In many reuse systems, a wide range of saline conditions can occur. For example, in fields at the end of the sequential reuse systems, soil salinities can be in excess of 40 dS m^{-1}. Reuse systems provide an opportunity to introduce crops with a wide range in salt tolerance, including very salt-tolerant crops that would otherwise not exist in the SJV. Some crops may be more appropriate than others, depending upon field conditions. Desirable and undesirable crop characteristics for selection under irrigation with saline water are listed in Table 11.2.

11.5.1 Vegetable Crops

The US Salinity Laboratory tested a number of novel leafy vegetable species to determine if they could fill a niche within a drainage water reuse sequence. Consistent with results from tests with most commercial vegetable cultivars, these were moderately sensitive to salinity, suggesting that they would perform best in the non-saline portion of a sequential reuse system. Purslane (*Portulaca oleracea*), on the other hand, grew well in sand tank experiments irrigated with simulated drainage effluent, suggesting that this crop may be suitable in saline portions of a sequence (Grieve and Suarez 1997).

11.5.2 Tree Crops

Eucalyptus plantations have been established throughout the SJV for the purpose of reducing the volume of drainage water that needs ultimate disposal (Cervinka 1994). Eucalyptus trees were proposed as an important component of the sequential reuse system (SJVDP 1990), but were not found to be as tolerant to frost or waterlogged soils as desired. They are commonly used in interceptor strips to reduce subsurface flows between the different stages of the sequential reuse system. Eucalyptus trees have been found to reduce dryland salinization in areas cleared of native vegetation in Western Australia (Bari and Schofield 1992). Re-vegetation with eucalyptus trees reduces groundwater recharge and lowers shallow saline water tables, thereby reducing salinization of the upper portion of the soil profile (Morris and Thomson 1983).

A study was conducted at the US Salinity Lab using large sand tanks to determine potential ET rates. *Eucalyptus camaldulensis* clone 4544 was irrigated with simulated drainage water that varied in salinity (EC 2–28 dS m^{-1}) and B concentration (1–30 mg L^{-1}) . Sand tanks were used because they drain well and the concentrations of salts and B in the soil water are close to those in the irrigation treatment water. When the average root zone salinity of the soil water was 15 dS m^{-1}, tree ET was reduced to 53 % of those in the non-saline treatment, when evaluated over the entire period that trees were subjected to saline water.

The reduced ET is attributed largely to salinity's effect on tree biomass. Eucalyptus tree biomass decreased as the soil water salinity increased above 6 dS m^{-1}. In addition, these investigators found a significant interaction between B and salinity (Grattan et al. 1996). At low salinity, increased B affected tree biomass, but it did not at higher salinity (i.e. equal to or $>$22 dS m^{-1}). Based on this sand-tank study, *E. camaldulensis* is classified more appropriately as moderately salt-tolerant, rather than salt-tolerant.

A field study testing the effectiveness of eucalyptus (*E. camaldulensis*, *clones* 4543, 4544 and 4573) was conducted in a hydrologically closed basin within the SJV. This district had fine-textured soils, primarily a clay to clay-loam. The trees were planted and irrigated with non-saline water to facilitate survival of the young trees. Trees were then irrigated with saline-sodic water (EC = 8.5 dS m^{-1}, SAR = 33) several months after they were established. The average EC_e and SAR in the 0–60 cm depth interval from 1996 through 1998 was 15 and 36 dS m^{-1}, respectively. Tree biomass was greatest in those plots that received gypsum applications. Fall-applied gypsum improved soil aeration, infiltration, and drainage during the winter when rain occurred. Substantial rainfall occurred in 1998 between Julian days 31 and 125 resulting in ponding in all treatments. Oxygen diffusion rates remained at 0 μg O_2 cm^{-2} min^{-1} in the untreated plot from Julian day 80–190; whereas, the rates were 0.3 μg O_2 cm^{-2} min^{-1} in the gypsum-treated plot after Julian day 140. Therefore, gypsum application substantially increased the oxygen diffusion rates in winter months and improved tree performance.

Pistachio is a very salt-tolerant nut crop. In a 9-year study, Sanden et al. (2004) evaluated the *Pistacia vera* 'Kerman' scion on four rootstocks under irrigation with saline drainage water at salinities ranging from 0.5 to 12 dS m^{-1} ECw (EC of irrigation water) and found no impact on yield with irrigation waters up to ECw = 8 dS m^{-1}. At ECw = 12 dS m^{-1}, nut yield was 81 % of the low-salinity control yield and the cumulative water use (ET) was 64 % of the control treatment. The salinity tolerance threshold for tested pistachio rootstocks ranged from 9 to 10 dS m^{-1} ECe. The authors cautioned readers that some deep roots of trees in high-salinity plots might have extracted non-saline water from adjacent low-salinity plots, even though plastic barriers were installed to a 1.5 m depth between treatments.

Athel (*Tamarisk aphylla*) and mesquite (*Prosopis alba*) are trees that can grow at considerably high salinities (ECe > 15 dS m^{-1}). In the sequential reuse system (IFDM) at Red Rock Ranch in the SJV, *P. alba* has been established in soils reaching 27 dS m^{-1} ECe. Although *T. aphylla* has little economic value, the wood of *P. alba* has very high value for furniture construction and charcoal production. In addition, the pods of *P. alba* contain 35 % sugar and 9 % protein that are very palatable for livestock (Velarde et al. 2003).

11.5.3 Salt-Tolerant Forages

A number of salt-tolerant forages may be useful in reducing drainage water volumes in the SJV and may also produce a food source for dairy cattle. Masters et al. (2007) point out that in spite of relatively little research to improve the feeding value of salt-tolerant plants through breeding or selection programs, several plants are capable of growing under saline conditions that could provide a feed source for livestock. At lower salinity levels, they include both legumes and some grass forages. At high salinity (>25 dS m^{-1}), production is substantially reduced and forage choices are fewer, but there are several halophytic grasses and shrubs that can produce 0.5–5.0 Mg ha^{-1} of edible dry matter per year. Fortunately, most research has shown that salinity does not lower forage quality (Suyama et al. 2007a) and in some cases, may slightly improve certain quality characteristics (Robinson et al. 2004; Masters et al. 2007). The exception may be the accumulation of certain minerals to excessive levels, particularly Se, Mo, and S, as discussed earlier, or the stimulation of secondary metabolites specific to plants growing in saline environments, which could be inhibitory or toxic (e.g. oxalate, coumarate, and nitrate) (Masters et al. 2007).

A greenhouse study was conducted in sand tanks at the US Salinity Laboratory in Riverside, California to evaluate the performance of ten promising grass and legume forages (Grattan et al. 2004a, b; Robinson et al. 2004). The legumes (alfalfa, *Medicago sativa* L. and trefoil, *Lotus glaber*) were sensitive to salinity but had good forage quality. The total biomass production of the alfalfa cultivars ('Salado' and 'SW 9720') was higher under saline conditions, than the other more salt-tolerant species. Tall wheatgrass (*Thinopyrum ponticum*, cv. 'Jose'), Bermuda grass (*Cynodon dactylon*, cv. 'Tifton') and paspalum (*Paspalum vaginatum* S., cv. 'PI 299042') emerged as the top choices, based on a combination of factors, including biomass production, salt tolerance, and overall forage quality (both organic and inorganic). Investigators cautioned that elevated tissue S concentrations could be problematic to ruminants relying on these forages as a sole source of feed (Grattan et al. 2004b).

Currently, field studies are underway to test the suitability of some salt-tolerant forages, such as wheat (*Triticum* spp.), forage brassicas, and safflower (*Carthamus tinctorius* L.), and forage-cropping strategies using saline-sodic water (Corwin et al. 2003, 2006; Kaffka et al. 2002, 2004; Suyama et al. 2007b). A 24-ha site with variable salinity in the western SJV was developed in 1999 to study the use of saline drainage and other wastewaters (ECw average was 3.6 dS m^{-1} with a range from 1.2 to 12 dS m^{-1}) for the production of forages and cattle (Kaffka et al. 2002, 2004; Corwin et al. 2003, 2006). Tile drains were installed at a depth of 1.3 m, 37 m apart, and detailed baseline soil assessment for soil physical and chemical properties was done before the project began (Corwin et al. 2003). Soils were characterized initially for chemical and physical properties, and survey work was done with GPS mapping. Soil sample site locations were determined using ESAP software (Lesch et al. 2000). A second survey was carried out in March 2003 and investigators found that the salinity and B in the top 60 cm of soil declined, while subsoil concentrations (60–120 cm) were unchanged (Corwin et al. 2006).

Bermudagrass (*Cynodon dactylon*, cv. 'Common' and 'Giant'), planted to eight paddocks each, remained productive (1.5 to 2.5 Mg $hectare^{-1}$ DW, depending upon cultivar) in areas where soil salinity in the 0–30 cm depth averaged 13 dS m^{-1} ECe after 5 years of irrigation with saline drainage and other waste waters (Kaffka et al. 2004). Livestock trials were carried out for 3 years (2001–2003), but grazing continued. Cattle were estimated to remove between 40 and 60 % of the standing forage biomass while grazing during the summer and a larger percentage by late fall (October) when grass growth rates declined. These fields supported beef cattle with weight gains of 0.75 kg day^{-1}, once Cu supplementation was administered to offset a deficiency, due in part to high S and Mo in the drainage water. In areas of the pasture where the soil salinities exceeded 20 dS m^{-1} ECe, the Bermudagrass did not grow well.

On a hay basis, forage crude protein (CP) contents averaged 9.0 %, (range: 4.2–22.1 %), acid detergent fiber (ADF): 29.6 % (range: 20.7–42.3), B: 245.4 mg kg^{-1} dry wt. (range: 73–1,004), Mo: 1.44 mg kg^{-1} dry wt. (range: 0.3–5.3) (Kaffka et al. 2004). Crude protein and trace element concentrations were greater in the upper portion of the canopy selected by cattle; Na content was greatest in the lower portion of the canopy. At this location, Mo, rather than Se is found in large amounts in some areas of the field. Generally, Cu:Mo ratios in ruminant diets below 3 or 4 are thought to affect cattle health but there is little formal research in this area (Suttle 1991). The Cu:Mo ratios at this site, however, averaged 5.2 and no Mo toxicity was observed during the study period.

In field studies conducted by Suyama et al. (2007a, b), tall wheatgrass cv. Jose emerged as a top performer among forages tested, due to its ability to maintain adequate dry matter yield (7.0 Mg ha^{-1} $year^{-1}$ or 2.9 t $acre^{-1}$ $year^{-1}$) and high forage quality (metabolizable energy of 9.3 MJ kg^{-1} DW) when growing in soils having ECe = 19 dS m^{-1}, SAR of 37, and B = 24 mg kg^{-1}. Creeping wild rye (*Leymus triticoides*) had higher dry matter yield (11.5 Mg ha^{-1} $year^{-1}$; 4.8 T $acre^{-1}$ $year^{-1}$) growing in a less saline field (13 dS m^{-1} ECe), but its forage quality was considerably lower (metabolizable energy of 8.1 MJ kg^{-1} DW). At this SJV location, Se, not Mo, concentrations are exceptionally high in the soils and drainage water. After multiple years of drainage water irrigation, these forages accumulated 6–11 mg kg^{-1} Se, well above the maximum tolerable concentrations (MTCs) of

2–5 mg Se kg^{-1} dry weight (NRC 1996, 2001). Such levels of Se could possibly cause toxicity in ruminants, if used as a sole forage source, but if fed at a rate of 20–40 g kg^{-1}, the forage could be used as a Se supplement in Se-deficient areas of the SJV.

11.5.4 Halophytes

Halophytes are defined as 'salt-loving' plants and their growth improves with increased salinity, up to a point. Among the halophytes, *Atriplex* spp. and to a lesser extent saltgrass (*Distichlis spicata*) are potential forages. In addition to 'Nypa Forage' saltgrass (Nypa Inc., Tucson, AZ), a native *D. spicata* variety has also performed well in a saline drainage water reuse system at a ranch in the southern SJV (Benes et al. 2004).

Salicornia (*Salicornia bigelovii*) and iodine bush (*Allenrolfea occidentalis*) have limited economic value at present, but they can thrive in soils with 50– 60 dS m^{-1} ECe in the top 30 cm of soil (Benes et al. 2004). Therefore, they both have a potential indirect economic benefit as crops that can further decrease drainage volume. Under irrigation with high Se drainage water (800– 1,200 μg Se L^{-1}), they can accumulate up to 14 and 7.5 mg Se kg^{-1} dry matter, respectively, which gives them potential as processed, organic Se supplements, commonly administered to livestock in Se-deficient areas. Salicornia can maintain ET rates in excess of reference evapotranspiration (ET_o) when irrigated with hypersaline drainage water (ECw = 29 dS m^{-1} and B >25 mg L^{-1}), according to Grattan et al. (1999). More remarkable is that this leafless plant loses the vast majority of its water as transpiration, not evaporation, from a wet soil surface (Grattan et al. 2008). Salicornia, which is one of the most salt-tolerant vascular plants, is sold as a salad supplement in Europe and produces oil in its seed that is equal in quantity and quality to soybean oil (Glenn et al. 1999). It has considerable promise as a halophyte for seawater irrigation (Glenn et al. 1998). Although this plant grows well vegetatively in the SJV, its seed production is considerably less than when grown along the coast.

11.6 Conclusions and New Research Directions

The reuse of saline-sodic water that contains high levels of trace elements (B, Se and Mo) has been successful in a number of field studies in the past 25 years, and management strategies have been evaluated using computer simulations. Various methods for reusing saline or saline-sodic water for irrigation include blending, cyclic use of two different water supplies that vary in salinity, and sequential reuse of water. The long-term success of irrigation with saline-sodic water, regardless of the strategy used, will depend on the evolution of practical management strategies that apply new research results and practical farming experiences, as they emerge in the future.

Irrigation with drainage water requires more careful management than irrigation with non-saline water. To be successful, salinity and soil physical conditions need to be monitored carefully over time to sustain crop yield and quality. Requirements include properly selected crops, providing the necessary leaching required to control soil salinity, and application of amendments to control the effects of sodicity on soil structure and infiltration. Trace elements should also be monitored to avoid accumulation in the soil and bioaccumulation within the crop, particularly in forage production systems.

Studies have shown that soil quality can be maintained, if sound irrigation and soil management are used, coupled with sufficient leaching for the crop grown. Slow infiltration of water and subsequent slow redistribution within soils are characteristic of many soils, particularly those with high clay contents, such as the salt-affected soils in low-lying areas of the western SJV of California. Adverse effects occur not primarily by the direct use of saline-sodic water for irrigation, but by subsequent rainfall or irrigation with low-salinity water. Irrigation with saline-sodic water followed by rain or irrigation with nonsaline-sodic water can enhance soil crusting, adversely affect the tilth of the seedbed, reduce seedling emergence, reduce infiltration rates, and aggravate waterlogged conditions, which can reduce soil aeration, thereby affecting crop growth. Researchers and growers have found many adverse affects to be mitigated by incorporation of gypsum or other amendments that liberate free Ca^{2+} in the upper portion of the soil profile.

Trace elements, such as B, Se and Mo, if present in the drainage water in high concentrations, can affect the extent to which this water can be used to irrigate certain crops. Particularly important is crop tolerance of B, or the accumulation of Se and/or Mo to levels hazardous for human or livestock nutrition. Boron toxicity has not been observed in many commercial (e.g. tomato, cotton, sugar beet, melon) or non-conventional crops (e.g. Salicornia) in drainage water reuse studies in California. Boron toxicity has been observed, however, on fruit trees, pistachio and Eucalyptus trees. On Eucalyptus and pistachio, some of this injury has been transient, and it is not clear to what extent the observed injury reduces tree growth. Some studies indicate that there is an interaction between salinity and B in many crops, whereby salinity reduces the toxic effects of B. The pH of the solution also plays an important role in salinity-B relations, but more research is needed to improve understanding of the mechanisms affecting ion interactions. Despite these encouraging observations, long-term effects of B accumulation in soils still remain uncertain, particularly if management practices are primarily focused on controlling soil salinity. Boron tends to be more resistant to leaching than salts. Consequently, it is possible that B concentrations in soils may continue to increase to levels that reduce yields of sensitive crops.

Although Se concentration was found to increase in crops that were irrigated with saline drainage water in the SJV of California, no reported concentrations in crops pose a health risk, even when irrigated with drainage water containing more than 300 μg Se L^{-1}. Although research is encouraging thus far, it remains unclear whether Se and/or Mo pose a potential threat over the long-term to ruminants fed with forages irrigated with saline-sodic water containing these constituents. Much depends on the type of livestock used, their stage of development, and the amount and type of forage in their diets. These issues warrant further study, and some work to address this concern is currently underway.

In addition, much more research is needed to identify appropriate crops in reuse systems. In particular, research is needed to identify appropriate salt-tolerant forages. Tall wheatgrass (cv. 'Jose') and Bermuda grass have both shown considerable promise in terms of salt tolerance, biomass production, and forage quality. In California's SJV there is a shortage of forages, and this trend will likely continue in the near future as dairies continue to increase, and pressure to remove beef cattle from foothill and mountain ecosystems increases.

Much has been learned about the use and reuse of saline-sodic water in the past three decades; however, more research is needed to continue this learning curve. Although the same set of scientific principles apply in all cases, there is not one management practice that uses saline-sodic water that will be appropriate in all areas, appropriate to every farmer, or farming operation. Rather, use and reuse of saline-sodic water will have to be customized to site-specific conditions to be sustainable.

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Chapter 12
Membrane Desalination of Agricultural Drainage Water

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12.1 Desalination as Terminal Treatment of Saline Drainage

The salinity of agricultural drainage water and groundwater in the San Joaquin Valley (SJV) is now about 3,000–30,000 mg L^{-1} total dissolved solids (TDS), which threatens to progressively diminish the agricultural productivity of the SJV (DWR 2000, 2003; Williams and Alemi 2002). Construction of a master drain discharging to the Sacramento-San Joaquin River Delta was halted in 1983 after the detection of high concentrations of selenium (Se) in the form of selenate ($SeO_4{}^{2-}$) ion found at Kesterson Reservoir, the site of a low-lying basin for collection of tile drainage in the West Central SJV (Hartshorn 1985). The resulting regulatory mandate to "solve the salt buildup problem in-basin" can, as illustrated in previous chapters, be only partially solved for farmland through irrigation management and saline water reuse. Continued collection, treatment, and containment of drainage in a hydrologically closed basin are required. Desalination is one potential alternative to large-scale evaporation ponds or other discharge sites (DWR 2003) currently in use, and management of Se bioaccumulation in ponds still remains a concern (DWR 2000; Gao et al. 2007). In addition, water evaporation from such ponds represents a net loss of water that could otherwise be potentially reclaimed and reused.

Desalting is a management option that can be used for drainage water volume reduction, while producing fresh water suitable for both agricultural and potable

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_12,

applications. Drainage volume reduction via desalination can significantly reduce the size of evaporation ponds needed to achieve zero liquid drainage discharge. Agricultural drainage water treatment via desalination, followed by evaporation of the desalination concentrate residual stream, reduces liquid drainage waste volume resulting in a solid phase which may be more manageable and may represent a valuable resource in terms of its mineral content. With desalination coupled with evaporation ponds to achieve zero liquid discharge, the problem of a net import of salts to the valley via various water projects and the continued mobilization of trace minerals in local soils become a more tractable challenge of management of solid residuals. Depending on the relative cost of water and power, desalting may provide a net economic benefit.

Reverse osmosis (RO) and nanofiltration (NF) membrane desalination can potentially offer a viable method to manage drainage water, producing economically valuable high quality water for either agricultural or urban reuse. Membrane desalting can be achieved at reasonably low pressures with excellent product water flux and reasonably high levels of salt rejection. However, high-salinity SJV drainage water often contains sparingly water-soluble mineral salts, such as gypsum ($CaSO_4 \cdot 2H_2O$), calcium carbonate ($CaCO_3$), and barite ($BaSO_4$). At economical water recovery levels for inland water desalting, the concentration of these sparingly soluble mineral salt ions on the feed-side and near the RO membrane surface can exceed the solubility limits of these and other mineral salts and consequently crystallize and deposit onto the membrane surface, forming a mineral surface scale, which can reduce the permeability of the membrane, potentially damage the membrane, reduce desalination efficiency, and increase the cost of the desalted product water. Another complication for desalination of SJV drainage water is that its salinity and composition with respect to sparingly soluble salts varies geographically and temporally and thus the resulting mineral scaling potential is dependent on water quality at specific location and specific time of the irrigation season.

In addition to mineral scaling, membranes can be fouled by particulate matter, microorganisms, and organic matter, again leading to membrane performance degradation. While various feed pretreatment technologies have been advanced in recent years for the removal of particulate, bacterial, and colloidal matter (Molseed et al. 1987; Brehant et al. 2002, Cote et al. 2001; Ebrahim et al. 2001; Gabelich et al. 2003; Kruithof et al. 1998; Hydranautics 2008; Paranjape et al. 2003; Taniguchi 1997; Wilf and Klinko 1999) mineral salt scaling remains the major impediment to successful implementation of high-recovery inland brackish water desalting. For brackish groundwater from the SJV (~3,000–30,000 mg L^{-1} TDS) and the relatively low salinity Colorado River (CR) water (~700–1,300 mg L^{-1} TDS), membrane RO desalination is typically limited to a product water recovery rate of about 50–60 % (Lee et al. 2003) and 80–90 %, respectively (Rahardianto et al. 2008; Gabelich et al. 2007), when using conventional means of fouling mitigation, such as coagulation and filtration, and using mineral scale controls, such as feed pH adjustment acid and antiscalants. Feed pretreatment increases the cost of RO desalination, while the variability of water quality (especially with respect to feed water scaling propensity) makes the stable operation of RO desalting a challenge.

12.2 History of Water Desalination in the San Joaquin Valley

Prior to installation of the first UCLA tubular RO membrane plant, several industrial firms conducted short-term desalination efforts (at Firebaugh in the SJV), using hollow fine fibers, plate & frame RO membrane units, and electrodialysis. Technical problems were encountered in each of these pilot facilities. The first SJV commercial RO plant was the UCLA tubular membrane plant, installed at Coalinga, California in 1965 to supplement the diminishing supply of drinking water in Coalinga (Loeb and Selover 1967; Glater 1998; Stevens and Loeb 1967; Loeb 1966), and operated continuously for an extended period of time with minimum maintenance. The plant, designed by J. W. McCutchan and S. Loeb at the UCLA School of Engineering and Applied Science, consisted of an array of 1-in. (2.54 cm) tubular cellulose acetate membranes inserted into titanium support tubes. Their then-newly developed RO membrane system provided a breakthrough in desalination technology by demonstrating the first production of potable water by RO. Following 3.5 years of supervision and research by the UCLA Water Desalination Research Group, the plant was turned over to the city of Coalinga (Johnson et al. 1969) and remained on-line, operated by the city of Coalinga for a total period of 7 years.

In 1971, the desalination group at UCLA designed a 1-in. (2.54 cm) tubular RO pilot plant to be installed at Firebaugh, California, which was similar in design to the Coalinga facility (Antoniuk and McCutchan 1973). Constructed at UCLA, the pilot plant operated from November 1971 through 1980. After 4 months of UCLA management and optimization of the facility (Smith et al. 1981), the plant was turned over to the California Department of Water Resources (DWR). Desalination efforts at Firebaugh were encouraging and provided an incentive for the DWR to explore further the feasibility of large-scale agricultural drainage water desalination to augment irrigation water supplies in the SJV (Smith et al. 1981). This led to the 1984 experimental Los Banos Desalting Facility in the central irrigation district of the valley (Molseed et al. 1987; Smith 1992). A unique feature of this test facility was construction of a stratified layer solar pond for energy production to provide power for the on-site desalination systems. Unfortunately, the Los Banos facility was closed in 1986 as a result of the Se-related order from the U.S. Environmental Protection Agency (EPA) to terminate tile drainage in the central area of the SJV. The closure of the Los Banos facility was a setback for development of large-scale capability for agricultural drainage water desalination in the SJV. In the last decade, research efforts in water desalination of SJV drainage water have intensified with ongoing pilot studies and efforts to develop cost effective systems for agricultural drainage water desalination.

Functioning in an integrated management program, desalination is potentially a local and/or regional-scale component for handling drainage water. Irrigation and drainage management at the farm level, including drainage water reuse, cannot fully address drainage. For example, evaporation ponds, which serve as a terminal treatment process, collect the unavoidable (but largely diminished) drainage water from properly irrigated and managed farm operations and further reduce the drainage volume that needs to be evaporated and then either sequestered or removed from the valley. Compared to the evaporation pond alternative, desalination offers the advantage of providing fresh water as a marketable product, although evaporation ponds themselves, while not providing additional fresh water supply, can also produce a potentially marketable mineral products.

In order to achieve the benefits that can be derived from desalination, one must address the technological challenges associated with regionally and temporally variable water quality. Accordingly, this chapter presents potential technology solutions and an evaluation of possible desalination process options.

12.3 Management Issues in Reverse Osmosis Membrane Desalting

12.3.1 Reverse Osmosis Technology

Reverse osmosis (RO) membrane technology utilizes a semi-permeable membrane that rejects dissolved solids, while allowing water to pass through the membrane. In the RO process, pressurized saline feed water flows along the membrane surface. The applied pressure provides the driving force for water permeation through the membrane. Salts are rejected by the membrane, and thus a high purity (low salinity) permeate stream is obtained on the permeate of the membrane. Most RO systems use a cross flow operation (Fig. 12.1) to reduce the salt concentration buildup near the membrane surface (i.e., concentration polarization). As the fluid permeate passes through the membrane, the portion of the feed that retains the mineral ions ("retentate") flows downstream. As the retentate flows along the membrane channel and its concentration (and thus osmotic pressure) increases, the driving force for permeate flux correspondingly decreases.

The transport of water across the membrane is governed by (a) the pressure applied to the feed water and retentate and (b) the osmotic pressure of the system. The water flux, J_v, across a RO membrane is a function of the applied trans-membrane and osmotic pressures as given by (Amjad 1993)

$$J_v = L_p \cdot (\Delta P_m - \sigma \cdot \Delta \pi) \tag{12.1}$$

in which L_p is the water permeation coefficient, ΔP_m is the trans-membrane pressure, and σ is the reflection coefficient. In general, σ, which varies from zero to unity, must be estimated, based on bench and/or pilot studies with single and multi-component solutions. In order to attain a net permeate flux, the applied trans-membrane pressure must be greater than the osmotic pressure

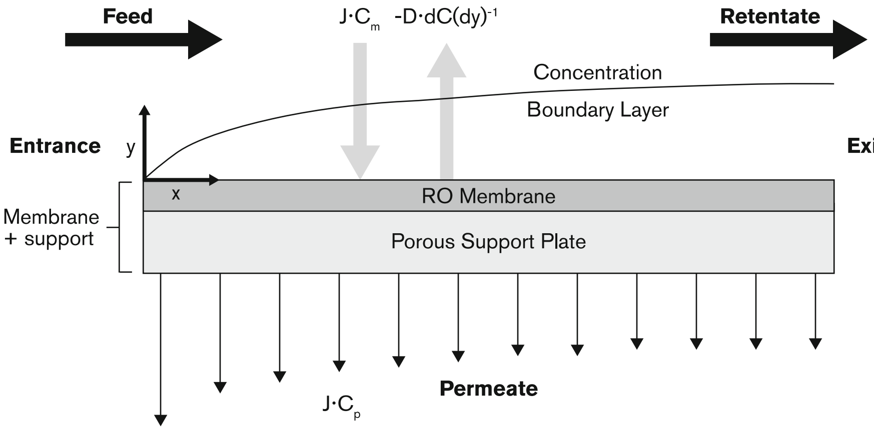

Fig. 12.1 Schematic of cross-flow plate-and-frame RO system showing the formation of a concentration boundary layer. J is the water flux, C_m and C_p are the respective concentrations at the membrane surface and in the permeate. D is the solute diffusivity, and dC/dy is the solute concentration gradient in the y-direction (McCool 2007)

difference ($\Delta\pi = \pi_f - \pi_p$, in which π_f and π_p are the feed-side and permeate-side osmotic pressures, respectively). The osmotic pressure, π, is defined as

$$\pi = -\frac{(\mu_1 - \mu_1^o)}{V_1} \tag{12.2}$$

where μ_1 and μ_1^o are the chemical potentials of the solvent in the solution and in its pure state, respectively, and V_1 is the solvent molar volume. For dilute solutions, the osmotic pressure can be estimated as

$$\pi = iCRT \tag{12.3}$$

where i is the van 't Hoff factor, C is the salt molar concentration, R is the ideal gas constant and T is the absolute temperature. For concentrated solutions and where non-idealities are important, the osmotic pressure may be calculated from more detailed thermodynamic expressions or with the use of available thermodynamic simulators (Rahardiano et al. 2008; OLI 2006).

The solute flux through the membrane, J_c, is described by the following expression:

$$J_c = J_v \cdot C_p = B \cdot \Delta C + (1 - \sigma) \cdot J_v \cdot C_{avg} \tag{12.4}$$

in which C_p is the permeate solute concentration, B is the solute permeation coefficient (typically empirically determined), $\Delta C = C_w - C_p$ is the difference in effective concentration, C_w is the solute concentration at membrane wall, and C_{avg} is the average concentration. At $\sigma = 1$, salt passage through the membrane is primarily by a diffusive transport mechanism, while for $\sigma < 1$, both convective and diffusive transport of salt across the membrane can occur (Spiegler and Kedem 1966) and for $\sigma = 0$ osmotic pressure does not limit solvent transport through the membrane.

As water permeates across a RO membrane, rejected salt ions accumulate near the membrane surface, resulting in the formation of a concentration boundary layer. Concentration of salts at the membrane surface can be approximated using the simple film model (Lyster and Cohen 2007):

$$\mathrm{CP} = \frac{C_m}{C_b} = (1 - R_o) + R_o \exp\left(\frac{J}{k}\right) \tag{12.5}$$

where C_m, C_b, and C_p are the solute concentrations at the membrane surface, in the bulk, and in the permeate, respectively, J is the permeate flux and k is the solute feed-side mass transfer coefficient, R_o is the observed fractional salt rejection ($R_s = 1 - C_p/C_b$), and CP is the concentration polarization modulus. CP increases along the RO membrane channel, reaching its highest value at the channel exit (Lyster and Cohen 2007; Zydney 1997). As the concentration and osmotic pressure at the membrane surface gradually increase (from the RO channel entrance to the exit), the effective net driving force for permeation decreases; thus, the permeate flux decreases towards the exit region (Fig. 12.1).

The retentate stream becomes more concentrated with increased recovery and is concentrated by a factor, CF, defined as (Rahardianto et al. 2008):

$$CF = \frac{C_C}{C_F} = \frac{1 - R_W(1 - R_S)}{1 - R_W} \tag{12.6}$$

where C_C and C_F are the respective concentrate and feed concentrations. The fractional salt rejection with respect to the feed concentration and the fractional permeate product recovery, R_w, are defined as

$$R_S = 1 - \frac{C_P}{C_F} \tag{12.7a}$$

$$R_W = \frac{Q_P}{Q_F} = 1 - \frac{Q_R}{Q_F} \tag{12.7b}$$

where C_P is permeate concentration and Q_P, Q_F, and Q_R are the permeate, feed, and retentate volumetric flow rates, respectively. As the RO feed flows through the membrane flow channel, the retentate stream is concentrated with the concentration factor rising rapidly, once the percent recovery begins to rise above about 80 %.

As the RO product water recovery (R_w) increases, the concentration of sparingly soluble mineral salts (e.g., calcium sulfate, calcium carbonate and barium sulfate) in the membrane channel can exceed their saturation limit, leading to mineral salt precipitation and thus membrane scaling. The degree of saturation is typically expressed in terms of the saturation index, which for calcite, gypsum, and barite, is defined as

$$
\begin{aligned}
SI_C &= (Ca^{2+})(CO_3{}^{2-})/K_{sp_{calcite}} \\
SI_G &= (Ca^{2+})(SO_4{}^{2-})/K_{sp_{gypsum}} \\
SI_B &= (Ba^{2+})(SO_4{}^{2-})/K_{sp_{barite}}
\end{aligned} \tag{12.8}
$$

where SI_C, SI_G and SI_G, are the saturation indices for calcite, gypsum, and barite, respectively, (Ca^{2+}), (Ba^{2+}), $(SO_4{}^{2-})$, and $(CO_3{}^{2-})$ are the activities of the calcium, barium, sulfate and carbonate ions, respectively, and $K_{sp_{calcite}}$, $K_{sp_{gypsum}}$ and $K_{sp_{barite}}$ are the solubility constants (products) for calcite, gypsum, and barite, respectively.

In desalting agricultural drainage water in the SJV via RO, the feed water typically contains high levels of sparingly soluble mineral salts, limiting the ability to operate the RO process to low water recovery levels (to avoid mineral scale buildup) and thus increasing the cost of water production. Under such conditions and when the feed has high propensity for membrane fouling by organic and colloidal foulants, the feed water supply requires pretreatment.

12.3.2 Reverse Osmosis Feed Pre-treatment

The primary objective of RO feed water pretreatment is to ensure that the RO membrane is not adversely affected by fouling, scaling, or chemical degradation. Materials that may cause fouling include suspended particulate matter, such as silt, clay, suspended solids, biological slime, algae, silica, and iron flocs that adhere to and accumulate on the membrane surface or even within the membrane matrix. Fouling typically occurs in the initial stages of feed and progresses gradually toward the end of the membrane. Scaling on the membrane surface by mineral salts typically appears first in the membrane elements towards the downstream end of the feed, where the retentate concentration is highest. Adequate feed pretreatment is essential in order to reduce the frequency of membrane cleaning and/or replacement, both of which add to the overall cost of water production. To make raw water compatible as RO feed water, the proper pretreatment of raw water must involve a total system approach for continuous, consistent, and reliable operation. The type and extent of a pretreatment system will depend on the type of source water, its chemistry, and the temporal variability of water quality. Typical guidelines for acceptable RO/NF feed water quality are illustrated in Table 12.1; drainage water in the SJV often exceeds these concentrations. The required feed water pretreatment must therefore be tailored, with respect to the desalting operating conditions (e.g., cross-flow velocity and recovery) and specific source water chemistry.

Table 12.1 Typical guidelines for maximum acceptable RO/NF feed water parameter

Parameters	Recommended maximum value
Turbidity	0.5 NTU
Total organic carbon (TOC)	2 mg L^{-1}
Iron	0.1 mg L^{-1}
Manganese	0.05 mg L^{-1}
Oil and grease	0.1 mg L^{-1}
Silt density index	3
Volatile organic carbon (VOC)	in the μg L^{-1} range
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Turbidity	0.5 NTU
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Oil and grease	0.1 mg L^{-1}
Silt density index	3
Volatile organic carbon (VOC)	in the μg L^{-1} range

Source: http://www.amtaorg.com/amta_media/pdfs/12_Pretreatment.pdf. Accessed April 2011

Table 12.2 Potential RO/NF chemical pretreatment options

Pretreatment option	Objective
Coagulants/polymers	Added as a part of coagulation/flocculation process to improve solids removal
Scale inhibitors	Sequester formed mineral salt nuclei and crystallites and adsorb onto crystal surfaces, thereby retarding the onset of crystallization and reducing the rate of crystal growth
Antifoulants	Keep foulants in suspension to prevent deposition and adsorption onto the membrane surface
Acids	Lower feed pH to increase the solubility of certain mineral salts (e.g., calcium carbonate)
Bisulfites	Dechlorination

Source: http://www.amtaorg.com/amta_media/pdfs/12_Pretreatment.pdf. Accessed April 2011

Raw RO feed water can be treated chemically and/or mechanically using a variety of approaches as summarized in Tables 12.2 and 12.3. Pretreatment is generally considered to be sufficient when the membrane cleaning frequency is limited to no more than about 3–4 times per year, and membrane elements last more than 5 years with stable water productivity and salt rejection within the design ranges. Frequent cleaning, which can obscure the impact of poor pretreatment, should not be adopted as a substitute for effective pretreatment practices. In selecting treatment strategies, one must carefully consider the feed water chemistry in order to avoid unexpected fouling due to interactions of feed treatment chemicals. For example, recent studies have shown that the use of coagulants can result in loss of antiscalant effectiveness, as well as an increased scaling and fouling (Gabelich et al. 2003, 2006; Shih et al. 2006).

Table 12.3 Potential RO/NF mechanical pretreatment options

Pretreatment option	Purpose	Reference
Pre-screens	Removal of large objects and sand removal	Pomerantz et al. (2008)
Cartridge filter	Protection of membrane elements	Molseed et al. (1987) and Shah et al. (2004)
Clarifier	Reduction of suspended solids	Molseed et al. (1987) and Qin et al. (2005)
Media filtration	Removal of suspended solids	Gabelich et al. (2003)
Activated carbon	Removal of organics and dechlorination	Kim et al. (2009)
Greensand filters	Reduction of iron/manganese reduction	Mott et al. (1993)
Ozone	Removal of organics and reduction of biological activity	Bruchet and Laine (2005)
UV	Reduction of biological activity	Aidan et al. (2007) and Bonnelye et al. (2005)
Full conventional plant (coagulation, flocculation, sedimentation and mediafiltration)	Removal of particulate matter and organics and reduction of biological activity	Gaid and Treal (2007) and Chuang et al. (2007)
Microfiltration/Ultrafiltration	Particulate and bacteria removal and organic reduction	Aidan et al. (2007), Chu and Li (2006), Ning and Troyer (2007) and Nandy et al. (2007)

Adapted from http://www.amtaorg.com/amta_media/pdfs/12_Pretreatment.pdf. Accessed April 2011

Chlorination to combat biofouling must be carefully monitored and applied in recommended doses to avoid premature membrane degradation from chlorine exposure. The effectiveness of antiscalants can depend on the specific choice of membranes and feed composition; thus, their selection and dosage must be carefully optimized (Shih et al. 2005; Rahardianto et al. 2006).

12.3.3 Mineral Scale Suppression with Antiscalants

Rosenstein (1936) reported the first record of chemicals being applied for calcium sulfate scale suppression. Subsequent studies on the use of chemical additives for inhibition of mineral salt crystal formation have demonstrated the efficiency of various polyelectrolytes on retardation of crystal growth (Liu and Nancollas 1975; Amjad 1985; Shih et al. 2004; Drak et al. 2000; Klepetsanis and Koutsoukos 1998; Hasson et al. 1998; Dalvi et al. 2000). During the past two decades, new generations of antiscalants (AS) have emerged commercially, in which the active ingredients are mostly proprietary mixtures of various molecular weight polycarboxylates and polyacrylates (Semiat et al. 2003; Al-Shammiri et al. 2000). Optimal molecular weights have been reported in the range of 1,000–3,500 Daltons

(Walinsky et al. 1983). Other polyelectrolytes including polyphosphonates and polyphosphates have also been applied successfully with certain types of feed waters (Klepetsanis and Koutsoukos 1998; Weijen and Rosmalen 1985).

The precise mechanism of scale inhibition is not clearly understood at this time, but it is generally accepted that AS may adsorb both onto the membrane surface and onto formed crystals or associate/complex with incipient nuclei (or crystals), and these phenomena may govern the inhibition of scale formation and growth (Oner et al. 1998). In supersaturated solutions of sparingly soluble salts, a significant delay in crystal nucleation and subsequent growth has been observed in response to AS treatment. This delay is referred to as the "induction-time" of the system, which occurs at remarkably low "threshold dosages" of AS on the order of 1–10 ppm. The scale inhibition capability of AS is related to chemical structure, molecular weight, active functional groups and solution pH–parameters that have been studied in depth by several investigators (Weijen and Rosmalen 1985; Tadros and Mayes 1979; Austin et al. 1975).

An important factor in determining the success of desalination of inland water is often the optimization of AS treatment with respect to type and dosage. Prior to field-testing or even laboratory studies on the performance of RO processes, it is important to identify the proper AS to use and the dosage-induction time relationship for the expected level of retentate supersaturation. Antiscalants can be ranked based on measurements of observed homogeneous crystallization induction time (Shih et al. 2004, 2005) for various solution conditions of interest (e.g., composition, pH, and temperature). Once candidate antiscalants are selected, one can then proceed with experimental membrane performance analysis to establish the optimal dosage requirement. Various methods of determining induction times for homogeneous crystallization of mineral salts, such as conductivity (Klepetsanis and Koutsoukos 1998), constant composition monitoring through pH control (Liu and Nancollas 1975; Weijen and Rosmalen 1985; Murabak 1998; Walinsky and Morton 1979), and light scattering through transmittance and absorbance studies (Shih et al. 2004; Hasson et al. 2001) have been proposed in the literature. Induction time studies have also been conducted whereby a membrane element is fouled in a RO system (Hasson et al. 2001; Lee and Lee 2000). More recently, a robust method was developed for evaluating dose effectiveness and mineral salt crystallization suppression capability of AS via homogeneous crystallization tests. In this approach, mineral precipitation is monitored *in-situ* (continuously) by a back-light scattering turbidity probe, with added monitoring of the dissolved calcium ion concentration to provide a confirmation of the observed induction time.

Antiscalant effectiveness for a number of commercial AS can be ranked, based on measurements of their observed crystallization induction time as illustrated in Fig. 12.2a for gypsum crystallization suppression. The induction time, t_{ind}, can be estimated from the turbidity-time curve (Fig. 12.2a) by fitting a line to the linear portion of the rapid crystallization region, and further extrapolating that line to the time axis. The point of intersection thereby identifies the induction time. The effectiveness of AS is signified by longer observed time for the onset of crystallization (Fig. 12.2a) and the efficiency of crystallization retardation (indicated by a longer induction time) increases with AS dosage (Fig. 12.2b).

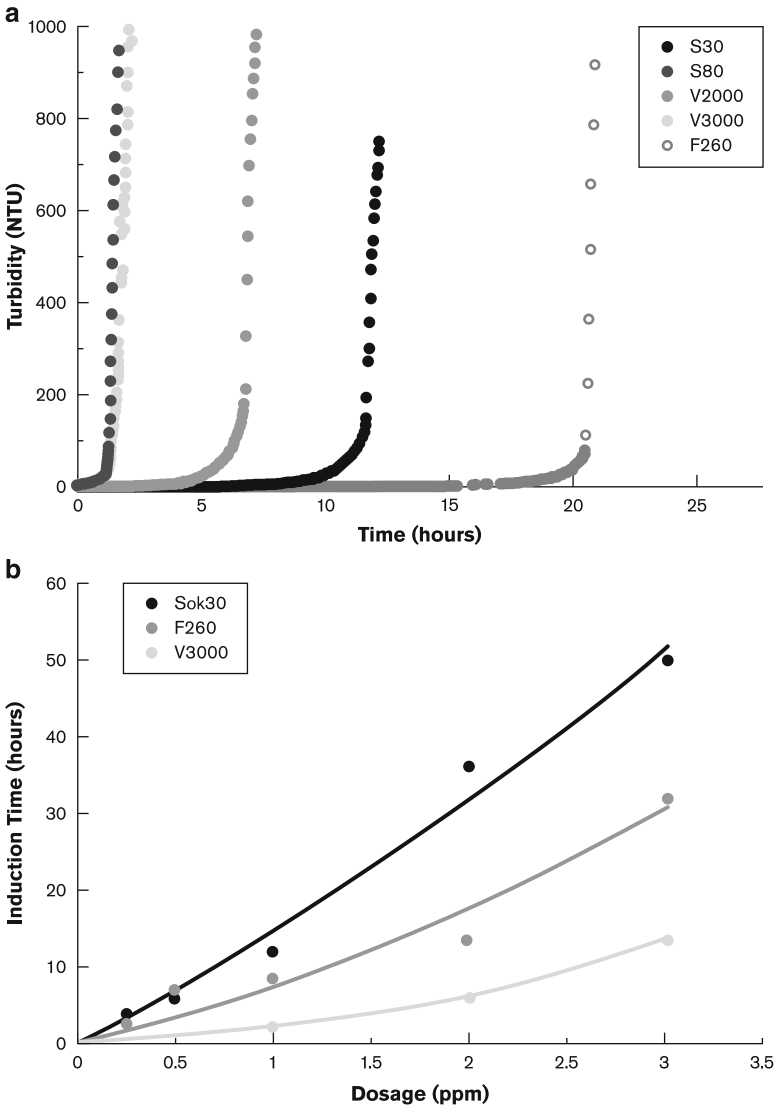


Fig. 12.2 (**a**) Turbidity plots for gypsum crystallization from model concentrate solution (TDS $= 26{,}250$ mg L^{-1}, $SI_g = 2.9$) with five different commercial antiscalants at dosage of 3 ppm including V2000 – a phosphino-carboxylic acid polymer; V3000 – a phosphonate-blend polymer; S30 and S80 – polyacrylic acids with molecular weight of 8,000 and 100,000, respectively, S30 is fully neutralized as the sodium salt of polyacrylic acid; and F260 – a polycarboxylic acid (Shih et al. 2004). (**b**) Dependence of crystallization induction time on antiscalant dosage. Solution and antiscalants are same as in Fig. 12.2a (Shih et al. 2004)

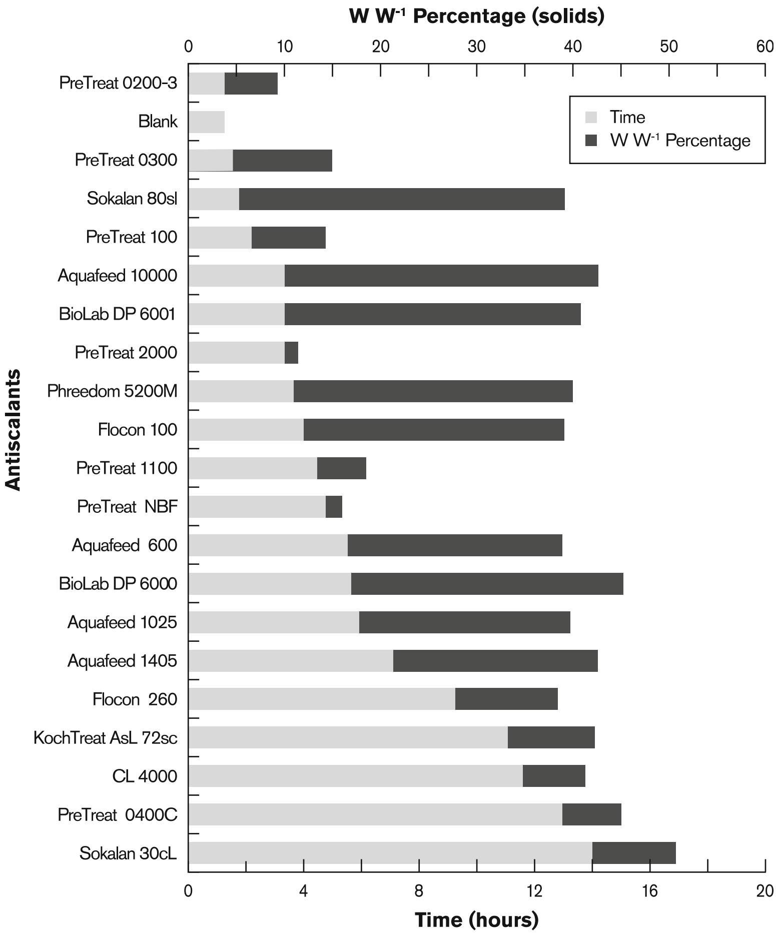


Fig. 12.3 Observed gypsum crystallization induction time for different antiscalants at a dose of 1 ppm. Also shown is residual solid content for the various antiscalant formulations. Solution composition corresponded to those of Fig. 12.2a

Antiscalants can vary significantly in their effectiveness (as quantified by the observed crystallization retardation time) and formulation (quantified in terms of their residual solids content) as illustrated in Fig. 12.3.

12.3.4 Membrane Selection

Membrane selection is typically dictated by the level of permeate flux and salt rejection desired; different membranes have distinct permeability, capabilities to reject various salts, and fouling resistance. The rejection of calcium (divalent, Ca^{2+}) and sodium (monovalent, Na^{+}) cations and membrane flux vary with membrane

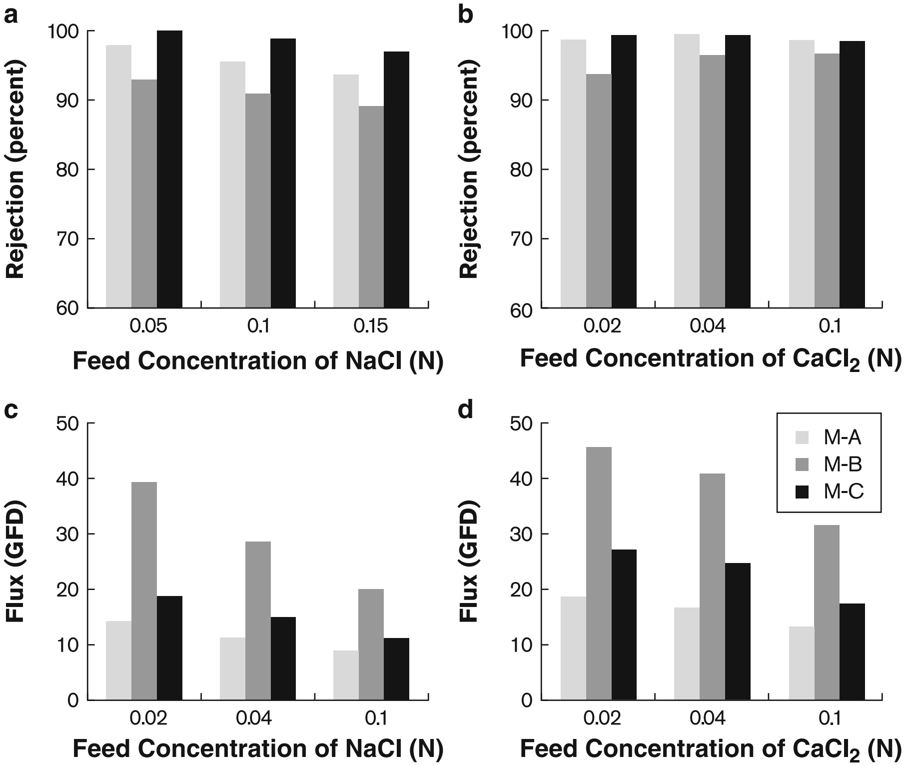


Fig. 12.4 Percent of rejection for candidate membranes at different feed concentrations: (**a**) NaCl and (**b**) $CaCl_2$ and flux for candidate membranes at different feed concentrations: (**c**) NaCl and (**d**) $CaCl_2$. Membranes are LFC-1 (M-A), TFC-ULP (M-B), and TFC-HR (M-C). Trans-membrane pressure = 1,379 kPa (200 psi) and temperature = 20 °C (Lee et al. 2003)

type, as illustrated in Fig. 12.5 for a number of low-pressure RO membranes. In general, membrane rejection is higher for divalent ions when compared to univalent ions (Lee et al. 2003). As expected, membrane rejection of sodium chloride decreases with increasing sodium chloride concentrations in the feed (Fig. 12.4a). This effect, however, is less apparent for calcium chloride solutions (Fig. 12.4b). As also expected, permeate flux decreases with increased salt concentration (Fig. 12.4c, d). For a specific desalting application, the membrane choice should be based on both performance (i.e., desired rejection and flux) and economic considerations (e.g., required membrane surface area).

Another factor, which may be of concern for membrane selection in the SJV, is the membrane biofouling resistance. The bacterial attachment assay is particularly useful for evaluating the biofouling potential of candidate membranes (Lee et al. 2003; Knoell et al. 1999). In this approach, membranes are contacted with a solution containing hydrophobic and hydrophilic bacteria strains that are radio-labeled with $Na_2{}^{35}SO_4$. After a prescribed equilibration period, the concentration of bacteria in

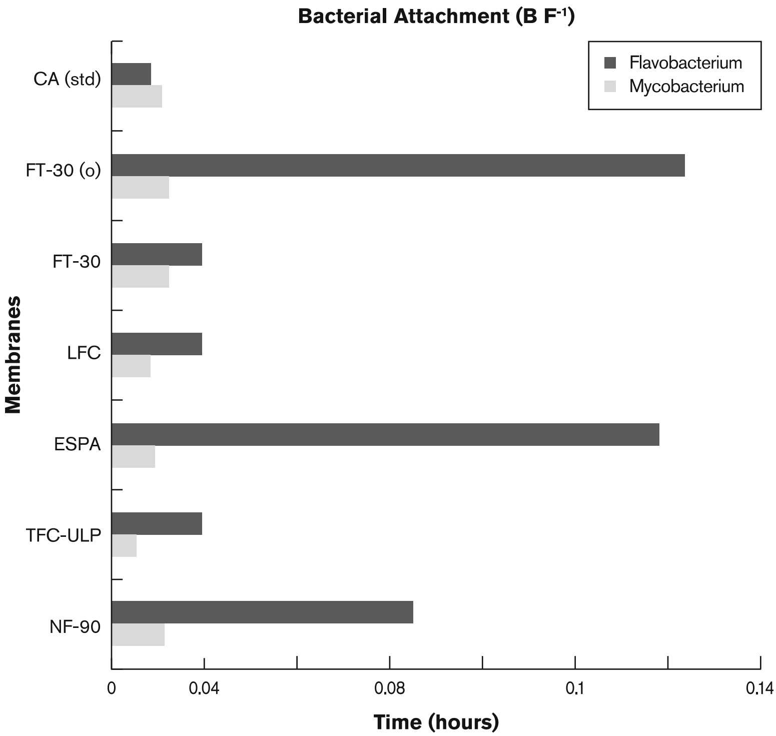


Fig. 12.5 Biofouling potential for candidate membranes in desalinating Buena Vista water without buffer among those CA (std), FT-30 (n), and FT-30 (o) are membranes used as controls. A hydrophobic strain of Mycobacterium and a hydrophilic strain of Flavobacterium were used as the test bacteria (Lee et al. 2003)

solution and the bacterial count on the membranes are determined and candidate membranes are compared, based on each membrane's bacterial attachment ratio (the ratio of the number of bacteria found on the membrane to the number of free bacteria in the solution). Commercial RO membranes have varying degrees of biofouling resistance, as shown in the example of Fig. 12.5 for cellulose acetate [CA(std), FT-30 (o) and FT-30(n)] and polyamide [LFC, ESPA, TFC-ULP and NF-90] membranes. Figure 12.5 shows that of the selected polyamide membranes, the LFC and TFC-ULP membranes were found to have the lowest biofouling resistance (i.e., lowest bacterial attachment ratio) of the tested polyamide membranes.

Recent studies have also shown that commercial membranes differ in their mineral scaling propensity, especially under the action of AS. The scaling propensity of RO membranes can be evaluated by carrying out scaling tests, such that membranes are compared at the same initial level of mineral salt supersaturation at the membrane surface (Rahardianto et al. 2006); this latter condition is achieved by ensuring that all comparisons are carried out at the same feed cross-flow velocity and initial permeate flux. Permeate flux decline data and the surface area of the membranes covered by scale are compared at the end of the test period (typically ~24 h) to quantify the mineral scaling propensity of the selected membranes. Higher flux decline and surface scale coverage are indicative of greater mineral scaling propensity. Recent studies have demonstrated that the membrane scaling propensity is different with and without the application of AS scale suppression. Therefore, membrane selection must also be optimized with respect to the membrane-antiscalant pair (Shih et al. 2005; Rahardianto et al. 2006).

12.3.5 Online Detection and Monitoring of Membrane Scaling

Early detection of membrane scaling and fouling is necessary for timely activation of preventive action. Traditional measures of RO plant performance trends (primarily flux decline and salt passage) are used as indirect indicators of the occurrence of mineral scaling and fouling (ASTM 2002; Saad 2004). Although numerous methods of scale and fouling detection have been proposed (Saad 2004; Mairal et al. 1999; Chen et al. 2004; Vrouwenvelder et al. 2006; Kang et al. 2004; Li et al. 1998, 2003; Mores and Davis 2001), it is only recently that real-time early detection of the onset of scale formation has become possible (Uchymiak et al. 2007). Direct visual observations and detection of mineral scale on RO membranes under high pressure have shown that it is feasible to utilize a transparent RO membrane monitoring cell along with digital image analysis to evaluate the kinetics of mineral scale formation (Uchymiak et al. 2007; Mores and Davis 2001). The use of a high-pressure flat sheet membrane cell, either with a transparent window (Uchymiak et al. 2007; Mores and Davis 2001) or a completely transparent RO cell (Rahardianto et al. 2008), allows real-time digital imaging of the membrane surface. For scale detection, the membrane monitor would typically receive a side-stream from a tail element of the RO plant where the retentate concentration is highest. The system can be adjusted, such that the level of solution supersaturation at the membrane monitor's membrane surface is at or higher than that for the last RO membrane module, thereby ensuring that scale would be detected first on the monitor's membrane surface. This type of scale detection adds an important monitoring capability to assess and optimize the efficiency of scale mitigation strategies and to ensure the onset of scaling is detected sufficiently early to enable effective scale mitigation.

12.4 Analysis of RO Feed Water Quality, Diagnostic Analysis of Permeates Recovery and Mineral Scaling

12.4.1 Analysis of SJV Agricultural Drainage Water Quality Data

The limitation that mineral salt scaling imposes on RO desalination of agricultural drainage (AD) water in the SJV can be assessed based on thermodynamic solubility

Table 12.4 Summary of annual average water quality in selected San Joaquin Valley locations for the most recent year of data (2003–2004)

Site	Location	TDS ($mg\ L^{-1}$)	pH	[Total carbonate][a] / [$SO4^{2-}$] ($mol\ mol^{-1}$)	SI_C[b] (20 °C)	SI_G[b] (20 °C)
CNR 0801	Southern Area, Kern Lake Bed	6,987	7.7	0.113	5.05	0.76
LNW 6467	Southern Area, Lost Hills	11,944	7.5	0.053	2.31	0.99
OAS 2548	Central Area	7,999	7.7	0.097	3.89	0.75
VGD 4406	Southern Area, Lemoore	23,480	7.9	0.048	5.29	0.84
ERR 8429	Southern Area, Corcoran	4,690	–	–	–	–

Source: McCool (2007)

[a][Total Carbonate] was calculated assuming HCO_3^- was the only species contributing to the total alkalinity

[b]SI_C and SI_G were calculated based on the water composition

– insufficient data

analyses, if given the compositional data of AD water throughout the SJV. The California Department of Water Resources (DWR) monitoring data for the period 1999 to 2004 (DWR 2003) from 55 sites provide a suitable database to explore the seasonal and geographical variability of water quality. Detailed multi-electrolyte solubility analysis (McCool 2007; Amjad 1993), performed for the various SJV sites, revealed significant geographical variability in water salinity and saturation levels with respect to the major mineral scalants of concern (i.e., gypsum, calcite, and barite), as illustrated in Table 12.4 for five selected source water locations for the 2003–2004 sampling period.

Agricultural drainage water quality can also vary significantly over the course of a year for each of the selected locations, as shown in Fig. 12.6. The water quality data do not reveal consistent seasonal variations or correlations between the gypsum saturation index (SI_G) and salinity (expressed as mg L^{-1} TDS). However, the SI_G follows changes in calcium concentration. The maximum absolute percent deviation of salinity (expressed as TDS) from the annual average values ranged from 12 to 52 % for the LNW and OAS sites, respectively. Salinity variations for the VGD and CNR sites deviated by as much as 40 and 30 %, respectively, about the annual average salinity for these sites. The average calcium ion concentrations ranged from 356 to 606 mg L^{-1} for the OAS and LNW sites, respectively, and did not correlate with the low or high average TDS values. The maximum absolute percent deviation of the calcium ion concentrations from the average values ranged from 7.4 to 37 % for the LNW and OAS sites, respectively. The sulfate ion concentrations do not correlate with salinity; the highest absolute percent deviation from the annual averages (about 50 %) was encountered at the OAS site. The maximum absolute percent deviation of the gypsum saturation index, SI_G, from the annual average values (for a given site) ranged from 5.2 to 45 % for the

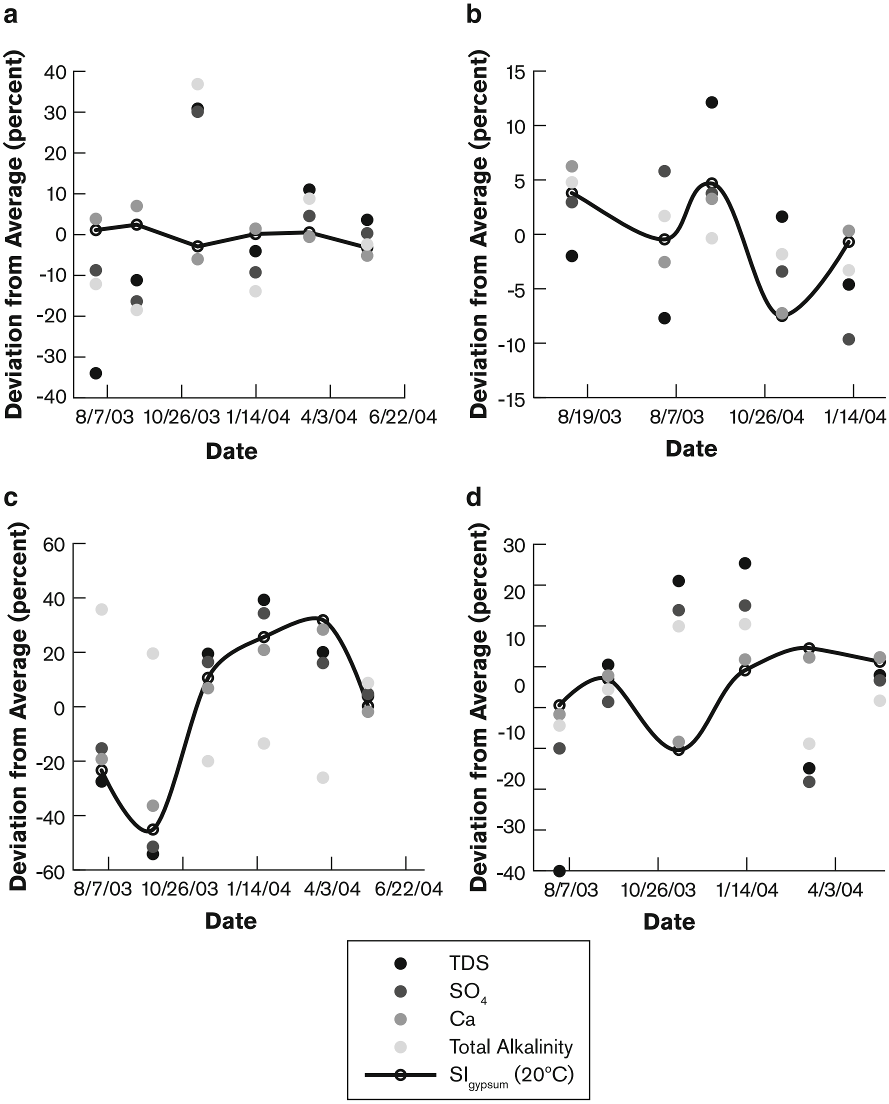


Fig. 12.6 Seasonal variability of water quality at sampling sites: (**a**) CNR, (**b**) LNW, (**c**) OAS, and (**d**) VGD

CNR and OAS sites, respectively. For the LNW6467 location, the gypsum saturation index, SI_G, was near or above saturation (0.92–10.4), in the range of 0.73–0.77 for the CNR0801 site, 0.4–0.99 for the OAS2548 site, and 0.81–0.92 for the VGD4406 site. Overall, the OAS site exhibited the greatest variability in water quality, while the least variability was encountered at the LNW site.

These variations in individual constituent water quality and the general lack of correlation among the water quality constituents have important implications for RO desalting, since current RO plants operate best when the quality of the feed water is consistent. Variation in water quality (i.e., TDS and saturation level with respect to various mineral salts) may require adjustments of trans-membrane pressure to cope with the change in osmotic pressure (Eq. 12.3). Increases in feed salinity and/or the concentration of scale ion precursors in the feed may also require reduction in permeate flow to reduce the level of concentration polarization (Eq. 12.5) and/or adjustment of scale mitigation strategies, such as the addition of AS to the feed (Shih et al. 2006; Rahardianto et al. 2006; Hasson et al. 1998; Murabak 1998; Ghafour 2003). If permeate production is impacted, this will also affect concentrate management. More importantly, plant design, in terms of the configuration and number of required membrane modules and thus, the surface area for permeation, may be drastically different, depending on the range of water feed quality. Therefore, large-scale implementation of desalination in the SJV would have to consider the following issues: (a) the use of smart desalination systems that can autonomously respond to temporal variations in feed water quality; and (b) distributed versus centralized desalination facilities to accommodate regional variability in water quality and associated concentrate management strategies.

In RO desalting, mineral scaling is likely to occur when the saturation indices of sparingly soluble mineral salts near or at the RO membrane surface exceed unity. Therefore, high concentration of minerals salts in the feed water will increase the likelihood for scaling during the RO desalting process. The solubility of mineral salt scalants is pH-dependent as is shown, for example, in Fig. 12.7 for the lowest and highest TDS conditions at the OAS and VGD sites, respectively. For the high salinity condition, the source water is nearly supersaturated with respect to gypsum for the VGD site (Fig. 12.7b), and supersaturated with respect to barite for both the high and low TDS conditions (Fig. 12.7). Among the scalants that are often of concern, calcite solubility increases with decreasing pH. Therefore, in RO processes, scaling by calcite can be mitigated by pH reduction of the feed water. Gypsum and barite saturation indices, however, are relatively pH insensitive and scaling by these salts cannot be managed by pH adjustment. Their precipitation can be inhibited by antiscalant addition to the RO feed stream. Scaling can generally be controlled for gypsum up to $SI_G = 2.3$. Barium levels in SJV AD water are reported at concentrations of less than 0.25–1.0 mg L^{-1}, which is essentially at about the detection limit for this cation. Barite tends to remain in solution, even at very high supersaturation levels, and its precipitation can typically be suppressed with AS up to $SI_B \sim 90$ (Hydranautics 2008). Silica scaling can also limit RO permeate productivity, but information on silica concentrations is incomplete for the 1999–2004 DWR monitoring database. More recent analysis of water quality for the five sites sampled suggests that silica concentrations in SJV AD water are in the range of about 23–43 mg L^{-1} (McCool 2007; Cohen and McCool 2007). At the RO recovery range of 50–75 %, for

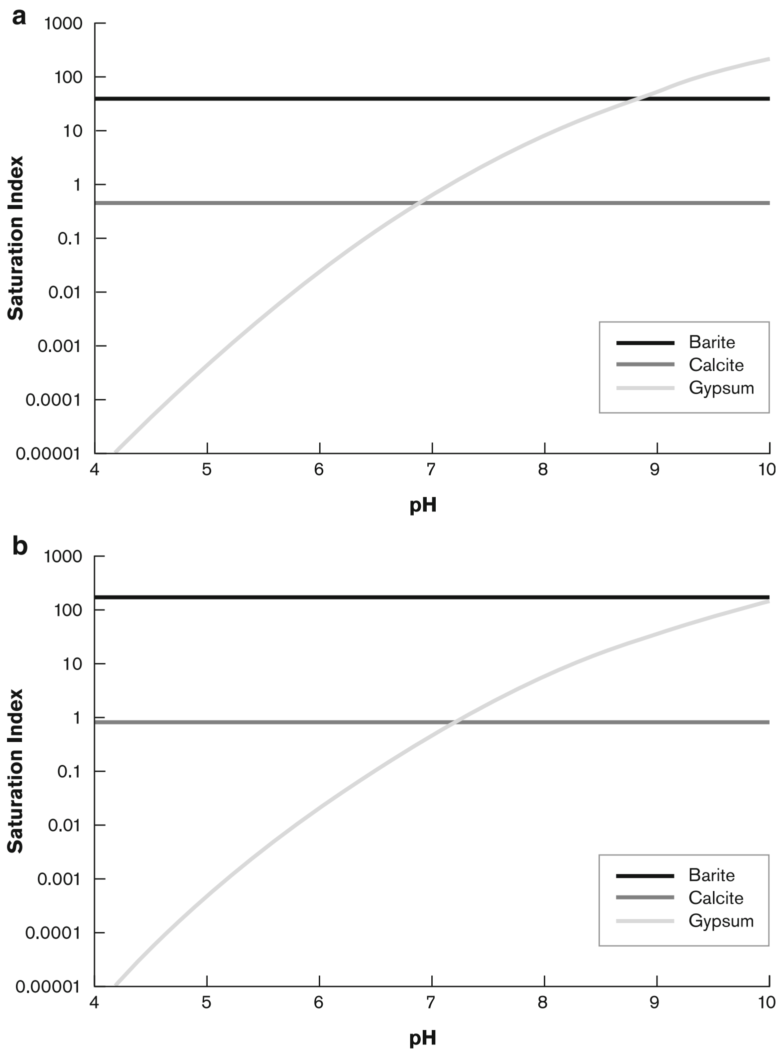


Fig. 12.7 Dependence of mineral salt solubility on pH: (**a**) OAS-2548, sample obtained at 09/09/2003, representing low TDS condition of 3,828 mg L^{-1} and (**b**) VGD-4406, sampled obtained at 01/13/2004, representing high TDS condition of 29,760 mg L^{-1} (McCool 2007)

example, the silica concentration in the RO concentrate, for the sites sampled, would be in the range of 46–86 mg L^{-1} (at 50 % recovery) to 92–172 mg L^{-1} (at 75 % recovery). It is reported that silica scaling can generally be controlled with AS when the feed or retentate silica concentrations are in the range of 160–240 ppm (McCool 2007).

12.5 Assessment of RO Recovery Limits Imposed by Mineral Salt Precipitation

12.5.1 RO Recovery Limits Based on Analysis of Water Quality Data

RO recovery limits imposed by mineral scaling can be estimated by calculating the recoveries at which the concentrations of sparingly soluble salts (e.g. calcite, gypsum, and barite) would reach their maximum controllable (i.e. non-scaling) supersaturation levels at the membrane surface. The saturation indices (Eq. 12.8), at a given recovery level, can be calculated based on the ion concentrations in the retentate stream that would result from operation at the specified recovery (Eqs. 12.5 and 12.6). The recovery limits can then be determined, based on either the requirement that the saturation indices for the scalants of concern do not exceed unity or that permeate recovery can be attained up to the limit of scale inhibition as enabled by either pH-reasonable adjustment (i.e., for calcite) or recommended AS dosing. In certain cases, the highest, tolerable trans-membrane pressure, which may be determined by economic or technical considerations, would dictate the highest attainable product water recovery. For example, RO membrane modules for typical brackish and seawater desalination may be rated for operation up to a maximum pressure of 4,138 KPa and 6,895 KPa (600 psi and 1,000 psi), respectively. The pressure constraints impose recovery limits, which are reached when the retentate's osmotic pressure approaches the above operational pressures limits.

Using detailed 2006–2007 water quality monitoring data (Table 12.5) for the five source water locations, the results of using this approach are provided in Tables 12.6 and 12.7 (McCool 2007). The saturation indices in these examples were determined using the OLI Systems Lab Analyzer 3.0 (OLI 2006), which predicts thermodynamic (including solubility) properties of mixed electrolyte aqueous systems. The range of salinity (4,115–28,780 mg L^{-1}) and saturation indices for this data set (Table 12.6) are similar to the 2003–2004 monitoring data (Table 12.4). With the exception of the ERR-8429 and the CNR-0801sites, gypsum saturation was near unity for three of the five locations. Silica content was also significant at a saturation index range of 0.22–0.34.

The saturation indices increase with product water recovery, as illustrated in Tables 12.6 and 12.7 for operation at both the natural source water pH and at pH = 6 (assuming 98 % salt rejection). RO desalting at pH ~6.0 is more typical of the low pH limit for calcite scale control (Fig. 12.7). Given that barium concentrations were at or below the detection limit, barite was not considered in the above examples. A summary of the recovery limits is given in Table 12.7. As recovery increases, the saturation indices for calcite, gypsum and silica all increase as shown in Tables 12.6 and 12.7. At the natural pH of the source water, the RO retentate stream would be oversaturated with respect to calcite

Table 12.5 Detailed water quality analyses, 2006–2007 (McCool 2007)

		Location				
Measurement	Units	CNR	LNW	OAS	VGD	ERR
Bicarbonate	mg L−1 $CaCO_3$	229	128	212	367	699
Boron	mg L^{-1}	13.5	17.5	23.5	43.4	2.6
Calcium	mg L^{-1}	350	625	462	422	88
Carbonate	mg L^{-1} $CaCO_3$	1*	1*	1*	1*	7
Chloride	mgL^{-1}	324	3,020	1,060	1,910	632
Fluoride	mg L^{-1}	5*	10*	10*	5*	5*
Hydroxide	mg L^{-1} $CaCO_3$	1*	1*	1*	1*	1*
Magnesium	mg L^{-1}	236	198	284	962	59
Nitrate	mg L^{-1}	344	155	46.7	51.9	51.3
Potassium	mg L^{-1}	46.7	5*	5*	7.8	3.5
Selenium	mg L^{-1}	0.032	0.223	0.18	0.05*	0.01
Silica	mg L^{-1}	23.5	37.9	31.4	43.2	38
Sodium	mg L^{-1}	1,250	2,820	2,780	9,270	1,250
Sulfate	mg L^{-1}	3,700	4,520	6,360	21,400	1,570
Total alkalinity[a]	mg L^{-1} $CaCO_3$	230	128	213	368	706
Aluminum[a]	mg L^{-1}	0.1*	0.1*	0.05*	0.5*	0.10
Arsenic[a]	mg L^{-1}	0.01*	0.014	0.006	0.05*	0.09
Barium[a]	mg L^{-1}	0.5*	0.5*	0.25*	2.5*	0.5*
Iron[a]	mg L^{-1}	0.17	0.152	0.045	1.41	0.28
Manganese[a]	mg L^{-1}	0.05*	0.05*	0.025*	1.55	0.59
Phosphorus[a]	mg L^{-1}	0.01*	0.03	0.08	0.12	1.96
Selenium[a]	mg L^{-1}	0.034	0.235	0.19	0.05*	0.013
Strontium[a]	mg L^{-1}	17.2	9.96	5.5	9.6	0.90
Total organic carbon	mg L^{-1} C	4.5	3.4	5.1	6.2	16.7
Dissolved organic carbon	mg L^{-1} C	4.2	4.6	5.1	6.2	15.8
Total dissolved solids	mg L^{-1}	6,372	11,270	11,020	28,780	4,115

Source: Smith (1992) and Uchymiak et al. (2007)
[a]Total concentration
*Value is at or below the reported detection limit

and thus RO desalting would not be possible without appropriate calcite scale suppression by either pH adjustment of the feed or use of AS. Although there is a slight variation in the saturation indices of gypsum and silica with pH, this variability is negligible, especially compared to that of calcite. In all cases, lowering the pH to about 6 would reduce the calcite saturation index below unity, up to a recovery of about 90 %. Once calcite scaling is suppressed (e.g., at pH = 6) and AS are utilized to reduce gypsum scaling, silica becomes the limiting scalant for the VGD and ERR sites, while gypsum is the limiting scalant for the CNR, OAS and LNW sites (Table 12.7). Recovery limits imposed by pressure limit constraints on the RO membrane vessels (Table 12.7) are higher than those imposed by the mineral scalants; therefore, they are not likely to be reached until complete scale suppression is attained.

Table 12.6 Water quality and saturation indices of calcite (SI_C), gypsum (SI_G), and silica (SI_S) for the 2006–2007 sampling data of selected San Joaquin Valley locations (McCool 2007)

Site	Sample date	Location	TDS (mg L^{-1})	pH	Total Alkalinity (mg L^{-1} $CaCO_3$)	SI_C	SI_G	SI_S
CNR 0801	7/31/2006	Southern Area, Kern Lakebed	6,372	7.5	230	2.70	0.70	0.22
LNW 6467	2/15/2006	Southern Area, Lost Hills	11,270	7.6	128	2.72	1.03	0.35
OAS 2548	4/10/2006	Central Area	11,020	7.6	213	3.02	0.99	0.29
VGD 4406	11/13/2006	Southern Area, Lemoore	28,780	7.6	368	2.18	0.95	0.38
ERR 8429	1/29/2007	Southern Area, Corcoran	4,115	8.0	706	9.50	0.12	0.34

Source: Cohen (2008)
Note: Values are those measured on the sampling date

12.5.2 Estimation of RO Recovery Limitations Based on Permeate Flux Decline Measurements

Experimental RO desalting tests are necessary to verify the occurrence of mineral scaling, and to verify the effectiveness of AS and pH adjustment in suppressing membrane scaling. Flux decline tests can be carried out conveniently using standard plate-and-frame RO systems, such as the one shown in Fig. 12.8 (Rahardianto et al. 2006). A summary of the percent flux decline, at the end of a 24-h RO test period for the different source water locations using the LFC-1 membrane (Hydranautics CA) is given in Table 12.8, listing the corresponding initial average saturation indices at the membrane surface, initial permeate flux, trans-membrane pressure, feed salinity, and pH (McCool 2007).

Contrary to conventional practice and assessment of scaling thresholds based on thermodynamic solubility analysis, RO operation at the natural source water pH would result in a lower flux decline relative to operation at lower feed pH. Recent studies have demonstrated that for AD water, which is high in sulfate and calcium (i.e., high gypsum scaling propensity) and lean in bicarbonate, RO operation at higher pH would result in suppression of calcite scaling by the sulfate ion and suppression of gypsum crystallization by the carbonate ion (Rahardianto et al. 2008). Product water recoveries are typically very small in small laboratory plate-and-frame RO units (typically below about 1–2 %). Therefore, the recovery level expected in a large-scale RO process must be estimated, based on Eq. 12.6, whereby the concentration factor (CF) is replaced by the average concentration polarization (CP_{avg}), divided by a concentration polarization allowance factor, a, which in a given spiral-wound element is typically in the range of 1.1–1.2,

Table 12.7 Field water sample recovery limits (Smith et al. 1992)

	Source water		Pressure recovery limit imposed by osmotic pressure		Recovery limit imposed by scaling				
					Calcite		Gypsum		Silica
Site	TDS (mg L^{-1})	pH	4,137 kPa (%)	6,895 kPa (%)	$SI_C = 1$ Natural pH	$SI_C = 1$ pH = 6 (%)	$SI_G = 1$ pH = 6 (%)	$SI_G = 2.3$ pH = 6 (%)	$SI_S = 1$ pH = 6 (%)
VGD 4406	28,780	7.6	69.9	81.4	Oversaturated	91.1	6.17	67.0	**54.9**
ERR 8429	4,116	8.0	94.3	96.5	Oversaturated	88.2	85.3	94.0	**63.4**
CNR 0801	6,372	7.5	94.3	96.5	Oversaturated	91.0	27.3	**68.2**	77.0
OAS 2548	11,020	7.6	88.9	93.1	Oversaturated	91.1	0.70	**57.9**	68.9
LNW 6467	11,270	7.6	83.2	89.6	Oversaturated	90.0	Oversaturated	**53.6**	62.1

Note: Bold values are the ultimate recovery limits with pH adjustment and antiscalant use

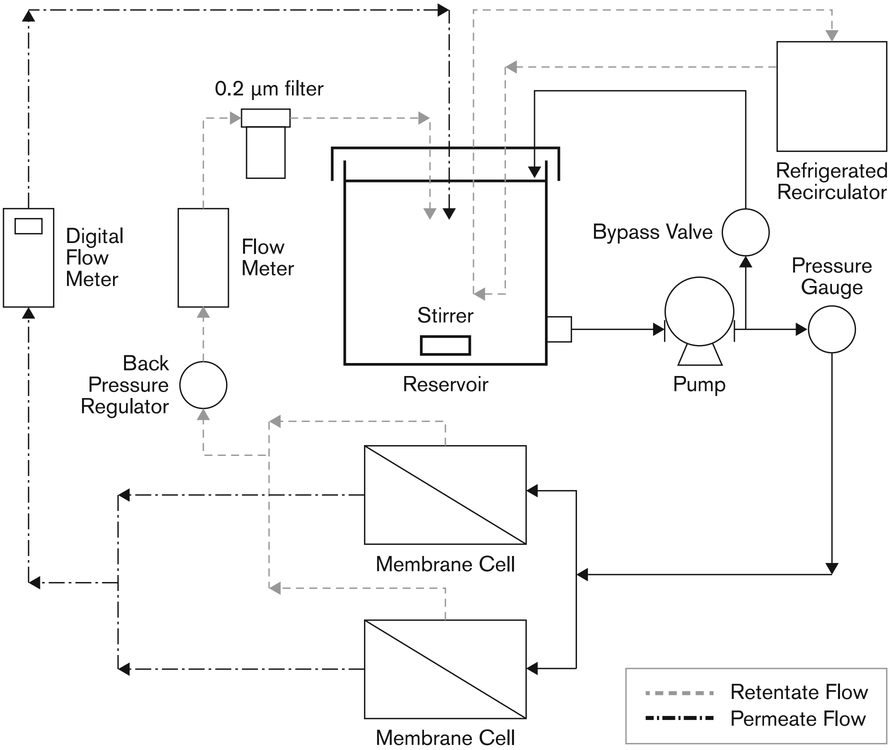


Fig. 12.8 Schematic of a plate-and-frame RO system

$$R_{W,eqv} = \frac{CP_{avg/a} - 1}{CP_{avg/a} - 1 + R_S} \qquad (12.9)$$

where $R_{W,eqv}$ is the equivalent product water recovery, CP_{avg} is the average concentration polarization modulus (Eq. 12.5) at the membrane surface, and R_s is the observed salt rejection in the given experiment. The above approach yields estimates of the recovery levels that would be attained when the average retentate concentration (at the exit module) from the RO plant would be equal to the solution concentration at the membrane surface in the laboratory plate-and-frame RO module. Following the above approach, the equivalent RO recoveries at the operating conditions of minimum scaling propensity (i.e., lowest flux decline) for the tested water sources (Table 12.8), were about 38 % for VGD; 53 % for ERR, CNR, and OAS; and 45 % for LNW. It is noted that, the above estimates are for the best conditions achieved with increased AS dosing and/or other scale mitigation options.

Table 12.8 Diagnostic flux decline experimental conditions and 24-h flux decline

Site	Condition	TDS (mg L^{-1})	pH	ΔPm (kPa)	CPavg	SIM,C	SIM,G	SIM,S	FD (%)	Rw, eqv (%)
VGD	Nat. pH	28,780	7.53	3,123	1.60	3.22	1.42	0.60	7.96	38
VGD	Low pH	28,780	5.56	2,979	1.59	0.01	1.43	0.71	8.88	38
VGD	Low pH & AS	28,780	5.93	3,137	1.61	0.04	1.44	0.72	9.36	38
ERR	Low pH & AS	4,116	6.00	1,903	2.12	0.13	0.30	0.79	7.54	53
ERR	Low pH	4116	5.37	2,110	2.12	0.01	0.31	0.79	11.6	53
ERR	Nat. pH	4,116	8.02	2,082	2.13	29.30	0.28	0.68	12.4	53
CNR	Nat. pH	6,375	7.50	1,917	2.11	7.37	1.53	0.46	6.37	53
CNR	Low pH	6,372	5.27	1,862	2.11	0.006	1.55	0.497	11.0	53
CNR	Low pH & AS	6,372	6.00	2,006	2.12	0.111	1.55	0.49	6.73	53
OAS	Low pH & AS	11,020	5.94	2,406	2.11	0.86	2.10	0.66	7.89	53
OAS	Low pH	11,020	5.47	2,579	2.1	0.014	2.10	0.66	47.4	53
OAS	Nat. pH	11,020	7.42	2,496	2.1	5.41	2.08	0.61	11.2	53
LNW	Low pH	11,270	6.48	1,951	1.8	0.37	1.94	0.68	14.2	45
LNW	Nat. pH	11,270	7.43	1,931	1.8	4.24	1.93	0.64	10.7	45

Sources: Smith (1992) and Uchymiak et al. (2007)

Note: AS antiscalant, *TDS* salinity measured as mg/L of total dissolved solids, *ΔPm* transmembrane pressure, *CPavg* – average initial concentration polarization modulus (Eq. 12.5), *SIM,C*, *SIM,G*, and *SIM,S* are the initial average saturation indices for calcite, gypsum and silica at the membrane surface, *FD* percent flux decline = $100 \times (1\text{-}F/F_0)$, where F and Fo are the permeate flux at time t and initially (t = 0); RW, eqv is the equivalent product water recovery in the RO plant (Eq. 12.9)

12.6 Reverse Osmosis Process Configuration: RO Stages

The common arrangement for RO desalting (Fig. 12.9) of brackish water (at recovery levels above 50 %) typically employs a 2:1 array. In this configuration, the retentate stream from the first stage, consisting of two parallel arrays of RO modules, is fed to a second RO stage of a single array of RO modules. The permeate product stream is the combined production from the first and second stages. The 2:1 array configuration is often employed to avoid exceeding the maximum recommended permeate flux in the first stage, while ensuring that a reasonable fluid cross-flow velocity is maintained in the second stage retentate stream. As a result of the increased osmotic pressure of the retentate stream, a booster pump may be required between the first and second stage in order to increase the applied transmembrane pressure to maintain the target permeate productivity.

This two stage configuration for RO desalination of SJV AD water was demonstrated in August-September 2000 in the Buena Vista Water Storage District in Buttonwillow, California to establish the feasibility of RO desalting (Lee et al. 2003). The plant consisted of a pretreatment multi-media filtration system and a two-stage RO system. A scale inhibitor (AS) was injected into the feed stream along with acid dosing to adjust the feed water pH to 6.8. This plant, designed to handle a feed water flow rate of up to 102 L day^{-1} (27 gallons day^{-1}), was in

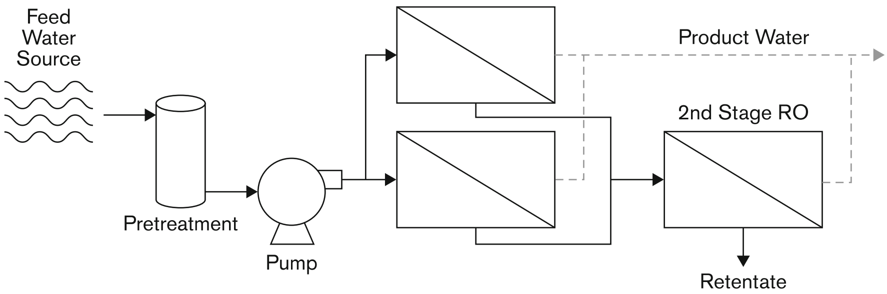


Fig. 12.9 Typical arrangement of a 2:1 array RO desalting process. The major percentage of permeate production is typically achieved in the first stage, while in the second stage the salinity is higher and thus permeate flux is typically lower. Various membrane module arrangements (e.g., with higher permeability second stage membranes) and a booster pump may be used between the first and second stage to increase the second stage pressure. Energy recovery devices can also be used (not shown) to recover energy from the high pressure retentate stream

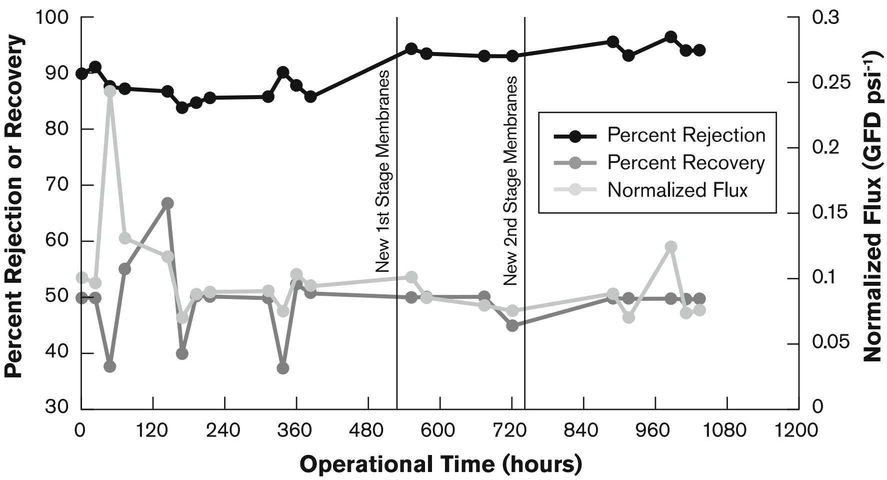


Fig. 12.10 Percent rejection based on measured conductivities, percent recovery, and normalized flux of the RO system during the operation of the pilot plant (Lee et al. 2003). Note: unit of right hand vertical axis 1GFD = 1 gal ft^{-2} d^{-1} = 1.66 L m^{-1} h^{-1} and 1 psi = 6.894 kPa

a 2:2:1:1 array (i.e., a 2:1 array with two serial modules in each array). The pilot plant, which operated during the period 8/1/2000-9/13/2000, experienced feed TDS variability over the range of 3,500–8,800 mg L^{-1}. Stable product water recovery of up to about 50 % was possible with periodic membrane cleaning (Fig. 12.10). However, higher product recovery would have necessitated scale mitigation strategies beyond standard approaches of AS addition and pH adjustment.

In addition to multi-stage (e.g., two-stage) RO configuration, RO arrays can additionally be arranged into multiple passes in order to improve the removal (rejection) of specific contaminants. Agricultural drainage (AD) waters in the SJV, for example, typically contain fairly high concentrations of B (Table 12.5; 2.6–43.4 mg B L^{-1}). Commercially available RO membranes have a relatively low B rejection (typically less than ~50 %) when desalting AD water at the native pH (pH 7−8), but can attain high B rejection of >95 % at highly alkaline pH (pH 10−11). To meet stringent product water quality requirements with respect to B (e.g., <1 mg L^{-1}), a two-pass RO configuration can be used. The primary, first-pass RO arrays (which can be a single-stage or a multi-stage RO configuration) would desalt AD water at the native pH in order to avoid membrane scale formation caused by mineral salts with highly pH-dependent solubility (e.g., carbonates, hydroxides, etc.). The permeate produced in the first-pass RO arrays would, however, contain a relatively high B concentration due to the low B rejection of RO membranes when desalting at low or near neutral pH. To further reduce the B concentration, the permeate from the first-pass RO arrays would be further desalted in second-pass RO arrays at elevated pH (pH 10–11). This will enable RO desalting with high B rejection, producing a final product water with the desired quality (with respect to B concentration).

12.7 Process Integrations for High Recovery RO Desalting

In desalting SJV AD water, conventional scale control methods (feed filtration, use of AS, pH adjustment), in addition to adjustment of permeate flux and cross-flow velocity (to reduce solution supersaturation at the membrane surface below the scaling threshold), can partly alleviate the water recovery limitations imposed by membrane mineral scaling. These conventional scale mitigation strategies can potentially enable RO desalination of SJV AD water up to a moderate range of water recovery levels (50–80 %). At moderate water recovery levels, however, the generated concentrate (i.e., brine) volumes remain large, likely making environmentally sustainable brine disposal methods uneconomical. Consequently, enhancing water recovery to high levels (85–95 %) is necessary to reduce the volume of high-salinity RO concentrate that must be discharged in, for example, a terminal evaporation pond. The critical challenge of water recovery enhancement can be addressed, in part, through process integration.

An example of process integration for enhancing water recovery is the integration of RO desalting with thermal evaporation processes (Ning 2002; Seigwirth el al. 1995; Rautenbach et al. 2000; Bond et al. 2003). Such integrated processes can achieve very high water recoveries (>95 %). While zero-liquid-discharge (ZLD) desalination processes are well established, their applicability for SJV AD water desalination is limited, due to their intensive energy demand. More promising alternatives integrate RO desalination with non-thermal unit operations that remove scale precursors (e.g., chemical precipitation, desupersaturation, ion exchange, and degasification) to enable overall recovery enhancement to high levels (85–95 %) at a reasonable cost. Such non-thermal process integration

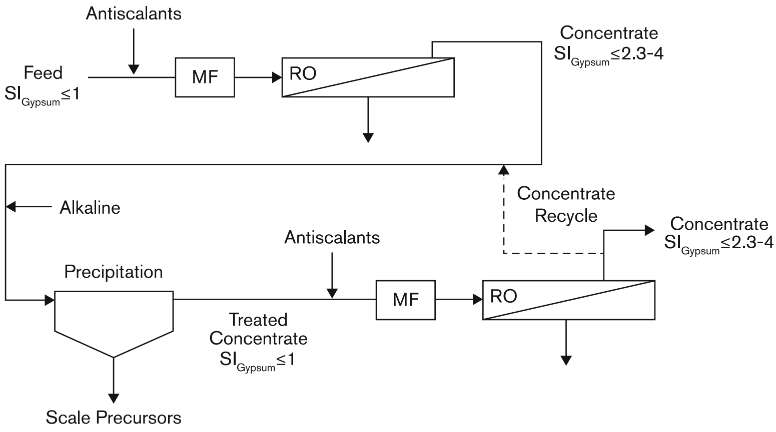


Fig. 12.11 A schematic for high recovery reverse osmosis (RO) desalination with microfiltration (MF) feed pretreatment and inter-stage precipitation process for reducing the concentration of gypsum scale precursors

(Fig. 12.11), for example, may be considered as a two-stage RO desalting process with an inter-stage unit operation that functions to remove scale precursors (Rahardianto 2009). With SJV AD feed waters that are typically near or at saturation level with respect to gypsum, feed water is desalted via first-stage RO up to a product recovery level dictated by the capabilities of conventional scale control methods. When the primary gypsum scale-control strategy for the primary RO stage consists of AS dosing, for example, $SI_G \sim 2.3$ to 4 is the typical range of upper recommended levels for the RO concentrate stream (Hydranautics 2008). Subsequent treatment of the primary RO concentrate via a scale-precursor-removal step would lower the SI_G to saturation (i.e., desupersaturation to $SI_G = 1$) or to undersaturation ($SI_G < 1$). Further product water recovery would become feasible through secondary RO desalting. It may also be possible, in certain cases, to further enhance the overall product water recovery via secondary RO concentrate recycling to the scale-precursor-removal stage, but this would be at the expense of additional feed handling capacity for the secondary RO unit.

In assessing the feasibility of a specific process integration approach for a given desalting application, thermodynamic calculations and process analysis can be very useful and revealing (e.g., Gabelich et al. 2007; Rahardianto 2009; Rahardianto et al. 2007). However, it should be recognized that membrane scaling is a kinetic process. Therefore, process feasibility should be further assessed through a systematic bench-scale experimental program and subsequent pilot scale studies. Such studies are needed to demonstrate that a proposed water recovery enhancement approach is technically and economically feasible for the removal of scale-precursors, with favorable kinetics. It is also important to demonstrate that a proposed process is effective for scale suppression during the desalting stage. The latter can be accomplished via a variety of scale monitoring approaches, including recently developed online visual monitoring of membrane scaling

(Uchymiak et al. 2007; Rahardianto et al. 2008). Moreover, membrane selection, AS choice and dosage, and other feed pretreatment strategies must be carefully optimized while considering the expected temporal variability of the source water quality to ensure stable and controllable desalting operations.

12.7.1 Chemical Precipitation

Reverse osmosis may also be combined with various chemical precipitation processes. A two-stage RO system with inter-stage accelerated chemical precipitation (ACP) of calcium carbonate as a scale-precursor-removal step, has been evaluated (Rahardianto et al. 2007) and pilot-tested (Gabelich et al. 2007) for the desalination of low-TDS Colorado River (CR) water (~700–1,000 mg L^{-1} TDS). The precipitation process is analogous to the classical lime-soda or caustic softening processes (i.e. precipitation softening) (AWWA 1999; Leowenthal et al. 1986). Alkaline dosing (e.g., pH adjustment with NaOH, lime, or soda ash) of the primary RO concentrate in a precipitation reactor (e.g., solids contact reactor) induces precipitation of primarily calcium carbonate, depleting the concentration of calcium in the aqueous phase, thus reducing the RO concentrate saturation index with respect to calcium-bearing mineral scalants (e.g., calcium carbonate and gypsum) to well below saturation. An added benefit is the co-precipitation of other mineral scale/fouling precursors, such as of Ba^{2+} (barium), Sr^{2+} (strontium), and silica, and also adsorptive removal of natural organic matter. Suspension of the precipitated solids is maintained in the reactor to provide preferential surface area for heterogeneous nucleation and growth of mineral salts, thus accelerating the kinetics of mineral precipitation and improving the efficiency of solid–liquid separation. Pilot testing showed that, at a precipitation reactor effluent pH above 10, the removal of Ca^{2+}, Ba^{2+}, Sr^{2+}, and silica can reach levels upward of 94, 97, 88, and 67 %, respectively (Gabelich et al. 2007). Permeate production with this approach demonstrated overall water recovery of up to 95 %, provided that good pH control was maintained in the precipitation reactor, along with AS to control silica scaling in the secondary RO step. It is noted that conventional precipitation softening is often characterized by the production of fine-suspension mineral salt crystals that generated sludge of low solids content (~2–30 %), requiring extensive dewatering (AWWA 1999). Alternative precipitation softening technologies have been developed to improve both precipitation kinetics and the efficiency of solid–liquid separation, including fluidized bed reactors and integrated precipitation-filtration systems (Graveland et al. 1983; Oren et al. 2001; Sluys et al. 1996).

The two-stage RO system with inter-stage chemical precipitation approach, developed and demonstrated for desalination of CR water, is applicable for desalination of SJV AD water (Rahardianto 2009), which is primarily limited by gypsum scaling. Due to the typically low carbonate alkalinity of SJV AD water; however, soda ash (Na_2CO_3) is needed to induce calcium carbonate precipitation and deplete the calcium concentration in order to significantly reduce the gypsum saturation

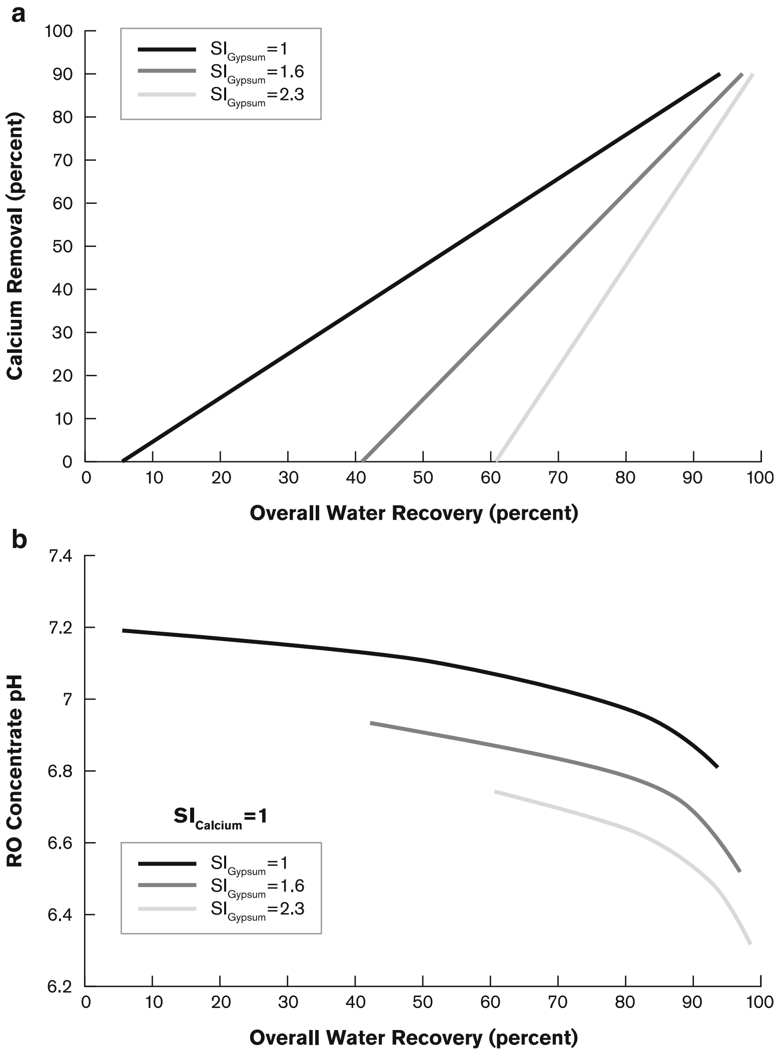


Fig. 12.12 Required calcium removal (as $CaCO_3$) during a chemical precipitation step that is integrated into RO desalination (as pretreatment or inter-stage treatment) and corresponding RO concentrate pH required to attain a target overall RO water recovery such that RO concentrate is (**a**) at a given level of saturation with respect to gypsum (SI_G) and (**b**) saturated with respect to calcite ($SI_C = 1$). Source water is LNW

index ($SI_G \ll 1$). Based on measured raw water feed composition, the method proposed by Rahardianto et al. (2007), using multi-electrolyte thermodynamic calculations, can be employed to assess the feasibility of the process approach for enhancing water recovery of RO desalination. Design plots, which are shown in Fig. 12.12a for the case of RO desalination of LNW source water, are generated by

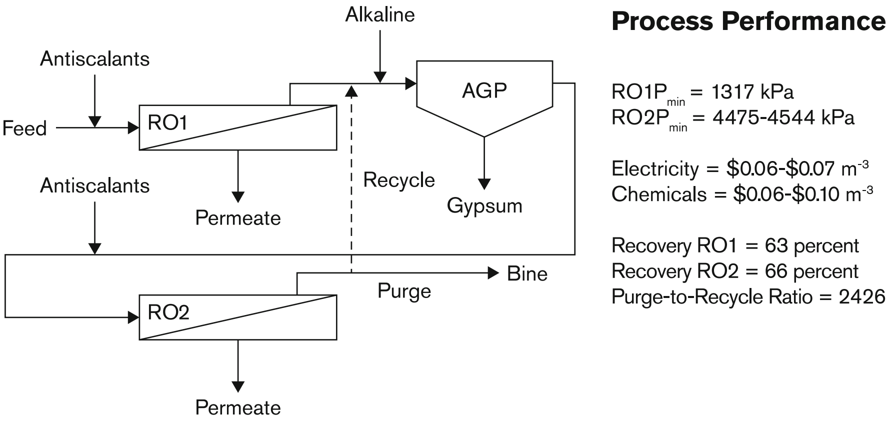


Fig. 12.13 Process simulation for high recovery desalting with AGP integration based on feed water of OAS 2548 (TDS = 9,600 mg L^{-1}), feed capacity of 1 MGD (22.84 m^3/s), permeate flux of 10 GFD (1.66 L h^{-1} m^{-2}), and overall recovery target of 95 % with permeate TDS of <500 mg L^{-1}

plotting the necessary levels of calcium removal in the chemical precipitation step (depleted as calcium carbonate) as a function of overall water recovery in order to maintain the desired SI_G in the RO concentrate. The overall water recovery and the desired SI_G limit also determine the RO concentrate pH which is necessary to keep the RO concentrate at the appropriate calcium carbonate saturation index value to avoid calcite scaling (Fig. 12.12b). Accordingly, the design plots (Fig. 12.12) can be utilized to determine the required chemical precipitation efficiency and feed-water pH adjustment to achieve a desired level of overall water recovery, based on selected criteria for the final RO concentrate saturation indices. For example, as shown in the design plots of Fig. 12.12, RO desalination of the LNW source water such that the concentrate is allowed to reach a SI_G of 2.3 (a typical recommended maximum level for effective mineral scale suppression with AS) would necessitate calcium removal of 73 % during the chemical precipitation step. Correspondingly, the RO concentrate should be kept at a pH of 6.5 to keep it just at saturation with respect to calcite.

12.7.2 *Accelerated Gypsum Precipitation*

A possible alternative to the integration of RO with an interstage accelerated chemical precipitation (ACP) is the integration of RO with accelerated gypsum precipitation (AGP) (Rahardianto 2009). Unlike ACP, which relies on the pH-sensitive precipitation of calcium carbonate to undersaturate the primary RO concentrate with respect to gypsum ($SI_G < 1$), AGP focuses on the desupersaturation of the primary RO concentrate ($SI_G \sim 1$) through accelerated precipitation of gypsum. As depicted in Fig. 12.13, the primary RO desalting functions to

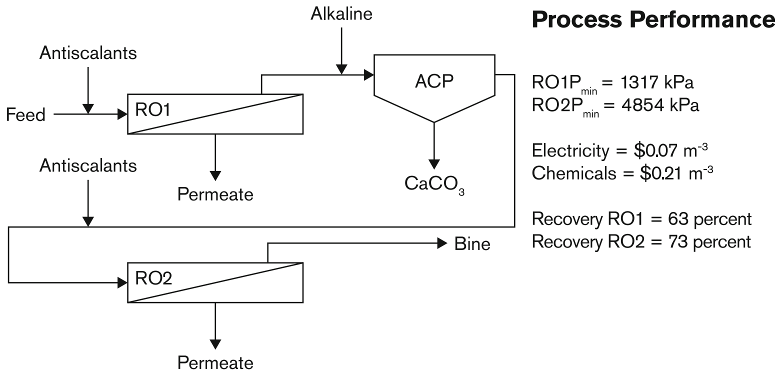
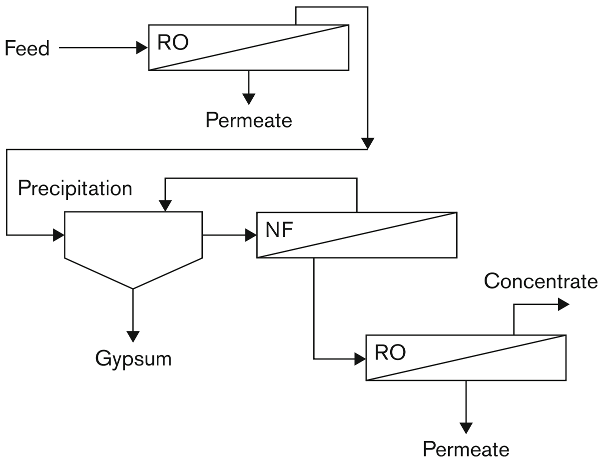


Fig. 12.14 Implementation of high recovery RO-ACP-RO agricultural drainage water desalination based on feed water of OAS 2548 (TDS = 9,600 mg L^{-1}), feed capacity of 1 MGD (22.84 m^3 s^{-1}), permeate flux of 10 GFD (1.66 L m^{-1} h^{-1}), and overall recovery target of 95 % with permeate TDS of <500 mg L^{-1}

generate gypsum supersaturation in the RO concentrate; subsequent addition of chemicals induces and accelerates the precipitation of gypsum and thus desupersaturates the primary RO concentrate, thereby allowing product water recovery in a secondary RO stage. In order to achieve a high level of overall water recovery, the RO-AGP-RO process may require partial recycling of the secondary RO concentrate stream to the AGP feed. This need for concentrate stream recycling reflects the role of AGP as a primary RO concentrate desupersaturator ($SI_G \sim 1$), as opposed to undersaturator ($SI_G < 1$), thereby limiting the attainable secondary RO recovery only to moderate levels. Nonetheless, the need for chemical addition in the AGP stage is limited to inducing and accelerating gypsum precipitation. Consequently, AGP integration with RO may potentially be chemically less intensive than ACP, which requires hydroxide and carbonate in order to drive calcium carbonate precipitation and thus reduce the calcium ion concentration. The presence of AS (for scale suppression during desalting) in the primary RO concentrate may complicate the ability to drive and accelerate gypsum precipitation in the AGP process. Therefore, effective means are necessary for accelerating the precipitation process, while reducing the sustained crystallization suppression of the AS added in the primary RO step (Rahardianto 2009).

The ACP and AGP processes have different costs, which must be considered when evaluating their practicality. Process analysis for the OAS water source (Table 12.5), using the RO-AGP-RO approach, demonstrated that up to 90 % recovery could be accomplished at a electrical energy cost of at least \$0.06–0.07 per m^3 of permeate (Fig. 12.14), assuming an electricity price of \$0.10 per kW-h and that kinetic energy in the concentrate streams of the primary RO (RO1) and secondary RO (RO2) steps are recovered using energy recovery devices at efficiencies of 95 % (Rahardianto 2009). Process analysis of the alternative RO-ACP-RO approach (Fig. 12.13) shows that the minimum energy cost is similar (\$0.07 per m^3 of permeate), but would require a higher cost of chemical additives (primarily AS and soda ash dosing in the APS)

Fig. 12.15 Integration of NF precipitation concentrator with RO membrane desalting



relative to the chemical cost (e.g., AS and lime dosing in the AGP) for the RO-AGP-RO process (Rahardianto 2009). The combined electrical energy and chemical cost for the RO-AGP-RO process are estimated at ~\$0.12-0.16 per m^3 of permeate product water compared to about ~\$0.28 per m^3 of permeate product water for the RO-ACP-RO process. The above cost estimates do not include capital cost. In addition, it is important to note that the overall cost would be dependent on the range of water compositions that would be experienced over the course of the desalting operation.

12.7.3 NF Precipitation-Concentrator

Another alternative integrated approach, previously demonstrated for desalting dumpsite leachate, is the treatment of high salinity feed with an auxiliary nano-filtration (NF) membrane concentrator system (Fig. 12.15) in which the NF concentrator is used for continuous recycling of the effluent from a precipitation reactor (Rautenbach and Linn 1996; Rautenbach and Woenkaul 2001). The main function of the concentrator is to increase the solution supersaturation with respect to gypsum (a major limiting scalant in SJV AD water) to a level in which kinetically-favorable gypsum precipitation can be sustained in the precipitation reactor. In addition, NF allows for the retention of scale precursors (within the scale-removal stage) and thereby enables secondary RO desalting of the NF permeate to enhance the overall product water recovery. Although the above process was reported to be capable of high overall desalination recovery (>95 %), it required intensive fouling control in the NF step by employing specially designed NF plate-and-frame modules, frequent NF flushing (every 30 s), and alkali cleaning (every 250–300 h).

A similar process also employing a NF concentrator system (DP3RO) has been proposed and tested for the treatment of SJV AD water (Enzweiler 2005), based on a concept of slurry precipitation and recycle RO (SPARRO), previously employed to treat mine drainage (Harris 1985; Juby and Schutte 2000). This process relies on the use of tubular NF membrane modules whereby seeded gypsum precipitation occurs in the retentate side (tube-side) of the NF membrane modules. Unfortunately, in the SPARRO process, premature membrane damage (e.g. due to scouring) may be encountered under various operating conditions (Seewoo et al. 2004, Juby and Schutte 1985). In addition, the use of tubular NF modules, as opposed to spiral-wound or plate-and-frame modules, requires significant plant footprint.

12.7.4 Other Process Integration Configurations for High Recovery Desalting

Process integration that does not involve mineral precipitation is also worth considering, such as integration of RO desalting with electrodialysis reversal (EDR) (Abulnour et al. 2003). EDR does not "concentrate" uncharged species; thus, water recovery limitations by non-charged mineral scalants, such as silica, are less significant in comparison to RO (ven der Hoek et al. 1998). Therefore, EDR can be particularly useful when integrated with RO for desalting AD water with high silica content. Another example, is the High Efficiency Reverse Osmosis (HERO) process that employs ion-exchange and degasification steps to remove calcium and carbon dioxide, respectively, prior to RO desalting (Jun et al. 2004). The above approach enables significant reduction of the scaling potentials of calcium-bearing minerals and allows RO operation at alkaline pH. The operation of RO at high pH can improve RO rejection performance (particularly with respect to B) and reduces colloidal and biological fouling (Jun et al. 2004). While the HERO process has been used successfully in commercial operation for ultrapure water applications, the process has not been evaluated for desalination of AD water. In recent years, there has also been increased effort in exploring emerging technologies, such as Membrane Distillation, Capacitive Deionization, Dewvaporation, and Forward Osmosis (Sethi et al. 2006). While none of these emerging technologies currently appear promising as a replacement to RO, there are ample incentive and potential for exploring various process integration options.

Over the last 30 years, there has been significant progress in RO desalination technology and in understanding the particular challenges for RO desalination of the various feed water from AD in the SJV. To enable widespread implementation of RO desalting processes for treating large volumes of AD water at distributed locations, the complexities of membrane scaling and fouling need to be addressed and integrated systems should be tested under site-specific field conditions. In an environment of increasing water demand and costs, RO desalting can potentially address both drainage and water supply problems in the SJV. In an integrated irrigation management scheme, RO desalting systems could be fed with a blend of AD water that has been substantially reduced in volume and somewhat concentrated via reuse. It would recover a significant portion of this feed drainage water as high quality product water that could augment fresh water supply in the SJV. At the same time, the remaining (concentrated) portion would have a significantly reduced volume, enabling discharge to a terminal evaporation pond or final management by other environmentally and economically feasible means (e.g., deep-well injection, processing with industrial thermal evaporators, etc.).

Acknowledgement Authors acknowledge the contribution of their colleague J. Glater.

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Chapter 13
Advances in Wetland Salinity Management

Nigel W.T. Quinn

13.1 Introduction

The Central Valley of California once boasted the largest contiguous natural wetland resource in the nation, but only 9 % of the wetlands that existed before 1850 remain. The California Department of Parks and Recreation estimates that California has lost 80 % of its salt marshes and 90 % of its freshwater marshes (Ferren et al. 1996). In the past, millions of acres of wetlands were seasonally flooded in the spring as snowmelt filled the San Joaquin and Sacramento Rivers, which overflowed their banks into the surrounding floodplain, dispersing seeds and germinating native grasses. The natural flood flows would inundate the bottom lands on either side of the rivers creating a vast region of natural grassland, which supported significant populations of migratory waterfowl and shorebirds that depended on the protein content of grass seeds and invertebrates to survive during the winter season. These wetlands were subsequently drained and the rivers that provided flood flows to them were dammed, permanently altering the San Joaquin River basin hydrology. Seasonal flooding was reduced and the remaining wetlands could support only a fraction of the historic migratory bird population during the winter months. Owners of wetlands and cattle ranches dependent on this seasonal hydrology began using surface irrigation return flows and subsurface tile drainage flows as a water supply, because of the diminished surface water supply available and because the higher economic return from agricultural crops out-competed wetlands for the available fresh water supply. The Kesterson crisis was actually a fortuitous event for the environment in that it gave acute recognition to a chronic problem that began when selenium-contaminated (Se) drainage water began being used as a wetland water supply.

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_13,

Subsequent restoration of a portion of the natural inundation flows by the federal government through its delta pumping facility at Tracy, California and the continued use of agricultural return flows from lands not affected by Se have helped to sustain about 68,797 ha (170,000 acres) of seasonal and permanent wetlands in the San Joaquin Valley (SJV). The legacy of the Se contamination at Kesterson Reservoir has been the requirement that wetland water supply quality be protected: water influent to the managed wetlands of the San Joaquin Basin cannot have Se concentration higher than 2 ppb (μg L^{-1}). There has been no similar water quality constraint placed on the salinity of influent water quality: meeting the Se constraint typically means that the salinity of the water supply remains below about 1,500 ppm total dissolved salts (TDS). Since seasonal wetlands must drain their salt loads entirely every spring, typically between March and May, improved wetland salinity management has more to do with improved scheduling of wetland drainage than a reduction in salt load export to the San Joaquin River. Real-time wetland salinity management is a strategy (to be elaborated later in the chapter) that offers an opportunity for wetlands to contribute to salt management in the San Joaquin Basin.

13.1.1 Definitions

Although this chapter focuses on managed wetlands, it is important to recognize other categories of wetland resources in California. These can be organized into three main categories, depending on their form and function:

13.1.1.1 Naturally-Occurring Wetlands

These resources are seasonal wetlands in the floodplain and vernal pools. These types of wetlands are rare in the Central Valley.

13.1.1.2 Ponds and Managed Wetlands

These resources are used for biological treatment of municipal and industrial waste. Ponds can be a very cost-effective and efficient means of treating agricultural liquid wastes. Wetlands are typically added to the treatment train as a polishing step prior to discharge to a receiving water body, such as the San Joaquin River.

13.1.1.3 Seasonally-Managed Wetlands

These are wetlands whose hydro period is dictated by management objectives that typically mimic the natural flood inundation cycle which, in the case of the San Joaquin River, existed prior to the construction of Friant Dam. Water is supplied by facilities, such as the Delta Mendota Canal, which conveys water from the

Sacramento – San Joaquin River Delta to local supply canals owned by agricultural and wetland water contractors according to a prescribed delivery pattern. Agency contractors, such as the San Luis and Delta Mendota Water Authority manage water deliveries for the US Bureau of Reclamation (USBR).

The focus of this chapter is on the management of salinity and Se in drainage water in the San Joaquin and Tulare Lake Basins to support wetlands and wildlife, with examples from an extended case study. In the San Joaquin Basin, the majority of wetland water supply, as previously described, is delivered from the Tracy pumping plant in the Delta. Lesser volumes are diverted from sloughs, such as Salt Slough, or obtained from high quality (Se concentration <2 ppb) surface return flows and operational spills from irrigation water districts, such as the Central California Irrigation District. In the case of the Tulare Lake Basin, the primary source of wetland water supply is low-Se, saline subsurface agricultural drainage water. Wetlands in this basin are more saline and designed and operated as evaporation basin mitigation. Hence, the discussion of the two seasonal wetland management systems implemented following the Kesterson crisis is partitioned separately within this paper into (i) duck clubs and wildlife refuges that support over-wintering waterfowl and whose purpose is to support conservation goals and recreation, and (ii) the assessment of risk to wildlife, as a result of exposure to evaporation basins, including measures designed to reduce and avoid impacts, and management of more saline wetlands, utilizing agricultural return flows to compensate for unavoidable impacts, and providing habitat benefits to nesting shorebirds and wintering waterfowl inhabiting the San Joaquin Valley.

13.2 Management of Seasonal Wetlands for Waterfowl Habitat

Management of seasonal wetlands in California's Central Valley is important for ensuring wildlife and habitat diversity, providing esthetic benefits, and facilitating recreational waterfowl hunting. The regional wetlands provide habitat for millions of waterfowl and shorebirds, a diverse community of moist-soil vegetation, and other common and endangered wildlife (Mason 1969; Small 1974; Cogswell 1977; Grassland Water District 1986; Shuford et al. 1998; Sibley 2000). Management practices that have evolved over time on these seasonal wetlands include grading, discing, mowing, grazing, burning, herbicide application, dry season irrigations, and the timing of wetland flood-up and drawdown. The fall flood-up occurs during the months of September and October, and the spring drawdown occurs during the months of February, March, and April. By timing flood-up and drawdown in the San Joaquin Valley, managers mimic the wet/dry seasonal cycle that these historic wetlands once experienced. Under "natural" conditions, this diversity would be supported through seasonal flooding and natural disturbances (drought, fire) that historically followed the seasonal cycle. This seasonal cycle can be adapted to promote desired species (Frederickson and Laubhan 1995).

Wetland "best management practices" (BMPs) focus on water level manipulation and are most often referred to as "moist-soil management", which means water level manipulations to promote productive habitat conditions and beneficial vegetation, such as smartweed (*Polygonum punctatum*), watergrass (*Echinochloa crusgalli*), and swamp timothy (*Heleochloa schoenoides*) for foraging waterfowl. During cool, wet years, and for wetlands of greater depth, managers generally drain the seasonal wetland later in the year because the optimal conditions of soil temperature and soil moisture tend to occur later. Conversely, during warm, dry years, and for shallower type wetlands, it is better to drain earlier. However, in intensively managed wetland complexes, such as the Grassland Water District, the heterogeneity of wetland soils, year-to-year variations in the weather, and the complex dynamic ecology of the wetland resource require constant hydrologic manipulation and fine-tuning of management decisions by wetland biologists. Altering wetland drainage schedules affects the timing and rate of drawdown of wetland ponds and hence the forage value of the wetlands for migrating and wintering shorebirds and waterfowl. The seeds of moist-soil plants are recognized as a critical waterfowl food source, providing essential nutrients and energy for wintering and migrating birds (Fredrickson and Taylor 1982; Bundy 1997; Shuford et al. 1998). Not only does the desirable vegetation provide direct nutritional value through consumption, but it also encourages healthy invertebrate populations, which are a high-protein food source at critical times of the year (Swanson 1988; Mushet et al. 1992; Smith et al. 1995; Bundy 1997; Stoddard and Associates 1998).

Three major moist-soil plant communities are generally targeted for waterfowl forage potential. These targeted communities are found in a mixed marsh setting and are either dominated by smartweed, swamp timothy, or watergrass. A healthy mixed marsh could include several other desirable species, such as sprangletop (*Leptochloa fascicularis*), brass buttons (*Cotula coronopifolia*), and alkali heath (*Frankenia grandifolia*). Several other acceptable plants work well in a mixed marsh community and can include, but are not limited to, tule or hardstem bulrush (*Scirpus acutus*), cattail (*Typha latifolia*), spikerush (*Haleocharis palustris*), purple ammannia (*Ammannia coccinea*), alkali bulrush (*Scirpus robustis*), fat-hen (*Atriplex patula*), and beggar-ticks (*Bidens* spp.).

Only in the last decade has applied research been undertaken to understand the role of water manipulation, irrigation, waterfowl habitat requirements, and both vegetation and waterbird responses to different management techniques. Results of this research, however, have not been effectively disseminated within the wetland community. Even though the majority of managed wetlands in the San Joaquin Basin are concentrated within the 68,799 ha (170,000 acres) Grasslands Ecological Area, these wetlands are independently managed by state, federal, and private entities. In the Grasslands Ecological Area, more than half of the area that is seasonally flooded is in private ownership: The Grassland Water District supplies water to 160 separate duck clubs, each of which has autonomy over the flooding and drainage cycles and moist-soil plant management within its boundaries.

13.3 Traditional Wetland Salinity Management

After the demise of the San Joaquin Valley Master Drain, the California Regional Water Quality Control Board (RWQCB) amended its San Joaquin Valley Basin Plan to promote a regional solution to the agricultural drainage problem, involving all contributors of salt within the basin to achieve compliance with water quality objectives. Resolution No. R5-2004-0108, passed by the RWQCB on September 10, 2004, further modified the Basin Plan to address persistent non-compliance with lower San Joaquin River water quality objectives that were not being addressed through voluntary adoption of irrigation and drainage BMPs. In this resolution, the RWQCB declared its intention to promote salinity management schemes, including timed discharge releases, real-time monitoring, and source control for all agricultural and wetland dischargers of salt to the river. These regulatory efforts affect seasonal wetland management and management of agricultural drainage.

In managed seasonal wetlands, salinity has always been of indirect concern in that it can affect the productivity and diversity of vegetation grown in the watershed (Rosenberg and Sillett 1991; Mushet et al. 1992). Salinity in wetland discharges during wetland drawdown is now also regulated to reduce downstream impacts of salinity and other contaminants in the San Joaquin River. Seasonal wetland drawdown, timed for optimal habitat conditions, occurs when salt-sensitive crops, such as beans, grown in the South Delta, are being irrigated for the first time and are germinating. High salt content in irrigation water diverted from the San Joaquin River or Delta channels connected to the river can negatively impact crop yields. Studies suggest that approximately 10 % of the San Joaquin River's annual flow and 30 % of its annual salt load pass through wetlands within the Grasslands Basin, which includes the Grassland Water District (Grober et al. 1995; Karkoski et al. 1995; Quinn et al. 1997; Quinn and Karkoski 1998). The salt load discharged by the north Grassland Water District occasionally contributes to exceedances of the assimilative capacity of the San Joaquin River, particularly during seasonal wetland drawdown. Figure 13.1 also shows that a drainage delay of between 2 and 4 weeks could help to improve water quality conditions in the river by shifting some of the excess salt load to periods of high assimilative capacity, such as the onset of reservoir flow releases to aid salmon migration. The Vernalis Adaptive Management Program (VAMP) provides significant flow releases for salmon migration down each of the major east-side tributaries to the San Joaquin River during the period April 15 – May 15 each year. The magnitude of flows and assimilative capacity provided are a function of hydrologic year (namely wet, normal, below average, dry or critically dry year). The management concept of changing the scheduling of drainage releases by either releasing earlier or later than traditional practices has been called "real-time wetland drawdown management."

To improve flow and water quality conditions in the San Joaquin River using a real time management approach, the California Department of Water Resources (DWR) formed the San Joaquin River Management Program (SJRMP), a stakeholder group representing many of the agencies, landowners, and other parties interested in improving the San Joaquin River ecosystem. One of the SJRMP's mandates was to reconcile and coordinate the various uses and competing interests along the river. The SJRMP

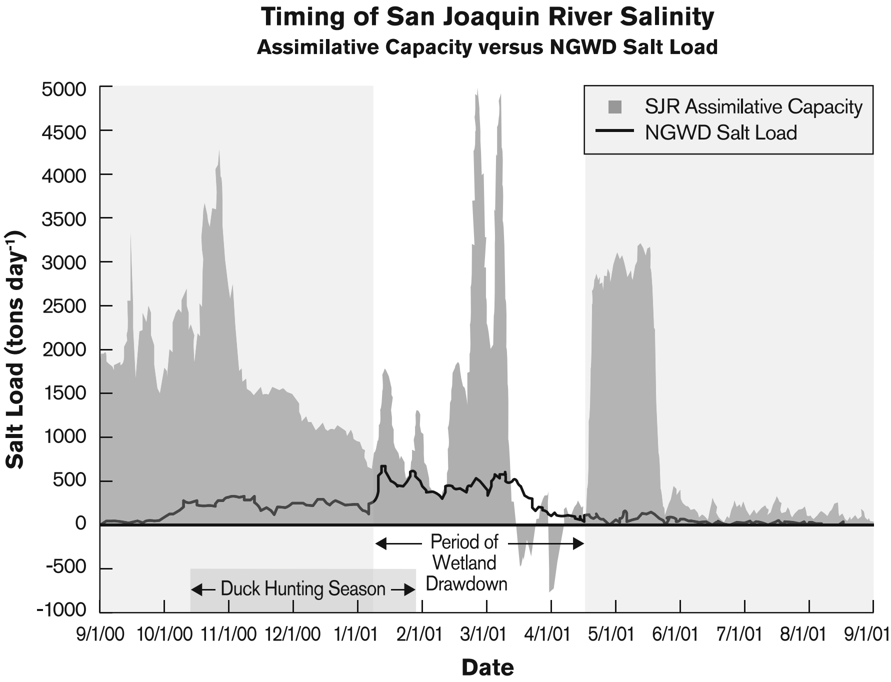

Fig. 13.1 Timing of wetland drawdown compared to the assimilative capacity of the San Joaquin River

created a number of working subcommittees, one of which was the Water Quality Subcommittee. This subcommittee applied for grants, one of which supported early work on real-time water quality management in the San Joaquin River, and it developed the concept of real-time drainage management to address the occurrence of high salinity levels in the lower San Joaquin River at critical times of the year, such as the onset of pre-irrigation on Delta agricultural lands.

Studies conducted initially under the SJRMP and subsequently by Lawrence Berkeley National Laboratory have suggested that wetland drainage from the Grasslands Water District could be scheduled to coincide with peak assimilative capacity in the San Joaquin River to help improve downstream water quality (Grober et al. 1995; Quinn et al. 1997; Quinn and Karkoski 1998). Increased surface water supply allocations under the Central Valley Project Improvement Act (CVPIA) have created greater opportunity than existed previously to coordinate the release of seasonal wetland drainage with the assimilative capacity of the San Joaquin River. Coordinated releases are intended to achieve salinity objectives and improve both downstream agricultural diversions and fish habitat in the mainstream of the San Joaquin River and Sacramento-San Joaquin Delta. Improved scheduling of west-side discharges can assist in avoiding conflicts at critical time periods for early-season irrigation as well as fish rearing and migration, removing an important stressor that can lead to improvements in the San Joaquin River fall-run Chinook salmon population.

13.4 Potential Impacts of Real-Time Wetland Salt Management

Wetland managers within the Grasslands Basin are concerned about the impacts of water quality regulations on the ecological functions in the wetland complex caused by soil salinization and the propagation of undesirable moist-soil plants if compliance with the newly promulgated regulation requires significant modification to wetland hydrology. As previously noted, changing the scheduling of wetland drainage to the San Joaquin River and the rate of drawdown of wetland ponds can negatively affect the forage value of wetlands for migrating and wintering shorebirds and waterfowl. Wetland salinity management could possibly also affect the productivity and diversity of vegetation grown in the watershed.

To address this issue, Lawrence Berkeley National Laboratory teamed with the Grassland Water District in 2000 to initiate what would eventually develop into a multidisciplinary research program to perform monitoring, develop field assessment technologies, and construct wetland water quality simulation models. Initially, telemetered flow and water quality monitoring stations were established at the district level (at the four major drainage outlet sites and the major inflow site for the northern area of Grassland Water District) and at the level of a single duck club (three drainage outlets and the main inflow site within the Salinas Duck Club). Second, new remote sensing techniques were developed using 2-m IKONOS imagery to attempt to discriminate the most important wetland moist-soil plant species and to use the spectral signals that were refined, based on ground truth surveys, to produce the first, remotely-surveyed images of the major wetland moist-soil plant associations in the Northern Grasslands Water District. Evaluation of this promising technique showed insufficient ground truth data were collected on this first series of experiments for proper accuracy assessment. A second study conducted on the San Luis Unit of the federal San Luis National Wildlife Area further refined the remote sensing techniques in comparison to the first project. A software program, e-Cognition, originally developed for medical imaging, was used in this second set of experiments to perform a segmentation analysis of the raw Quickbird 2-m imagery. Image segmentation is a technology that builds objects from the original raw pixel data, based on specified rules and a number of spectral and textural attributes. The clusters of objects were subsequently analyzed using ERDAS-Imagine software and the maximum likelihood classifier, a more conventional software tool for vegetation classification. Use of e-Cognition greatly enhanced the overall accuracy of image classification based on ground truth data. Accuracies of 65 % were achieved for 28 moist-soil plant associations and more than 90 % for plant associations of swamp timothy (*Heleochloa schoenoides*).

Research projects to assess the long-term viability of wetland salinity management were initiated by grants from the State Water Resources Control Board (SWRCB), the UC Salinity/Drainage Program and the DWR. The objective of this pilot project was to implement real-time wetland drawdown management on six pairs of wetland sites, both within the Los Banos Wildlife Management Area and the Grassland Water District, and to assess the environmental impact of this modified wetland hydrology practice. Each paired wetland site comprised a traditionally-managed wetland and an adjacent wetland where wetland drawdown was held to coincide with VAMP flows in the San Joaquin River, intended to improve survival of downstream migrating juvenile Chinook salmon, which also greatly increases the river's assimilative capacity for salt loading. Inflow and outflow at each site was monitored, as well as water temperature and salinity. Surveys of vegetation using remote sensing with appropriate ground-truthing were performed twice each year to correspond with the two dates when aerial imagery was flown. A comprehensive biological monitoring program was undertaken by the California Department of Fish and Game (CDFG) with internal grant funding and funds from the CALFED Ecosystem Restoration Program. This monitoring program included additional vegetation survey sites, estimation of swamp timothy seed head production, and visual surveys of waterfowl use in the project wetlands.

For each of the wetland salinity management projects previously described, continuous flow and EC monitoring stations were installed within wetland channels (Grassland Water District), at major drainage outlets, and at inlets and outlets of wetland impoundments within the project study area. Over the 7 years that these projects have been conducted, flow and water quality monitoring technology has improved dramatically. Initial flow measurements conducted at the Salinas Duck Club provided good water quality data; however, the acoustic Doppler sensors used to estimate flow lacked sensitivity at the very low flows experienced within the drainage outlets. Acoustic Doppler sensors currently used in the real-time drawdown management implementation project (Mace Agriflow units) demonstrate a ten-fold improvement in sensitivity and are capable of reading accurately to 0.003 $m^3\ s^{-1}$ (0.1 $ft^3\ s^{-1}$).

Finally, field assessments of soil salinity have been conducted since 2001 on various study sites within the Grassland Ecological Area. The first surveys were conducted by Fresno State University using a Geonics DD EM-38 instrument and a torpedo sensing system that was adapted for use on wetland soils. The motorized torpedo-dragged sensing system was replaced by a walking survey in later soil salinity assessments to avoid damage caused by the ATV on seed heads that were important to the biological surveys.

Work to develop a system for real-time management of seasonal wetland drainage, while providing for appropriate plant and invertebrate forage communities at seasonal wetlands, is ongoing but demonstrates (a) the shift away from an *ad hoc* management focused on a single objective (waterfowl) towards multi-objective management and (b) the commitment to enhancing the tools for accomplishing this more sophisticated and data-driven management strategy.

13.5 Drainage Management to Reduce Impacts to Seasonal Quality Fluctuations in the Grasslands Basin

The concern about Se bioaccumulation in drainage from agriculture in the Grasslands Basin is another constraint on drainage, and efforts to address it illustrate another critical aspect of "new perspectives" on management. Quinn and Karkoski (1998) describes the multi-agency response to integrated management of drainage to avoid wildlife impacts. The program, developed to avoid discharging drainage that would exceed the assimilative capacity of the San Joaquin River, bypasses drainage around sensitive waterfowl seasonal wetlands, and ensures timely deliveries of low-Se water to wildlife refuges, first required by the concerted action of six agricultural water districts, Grassland Water District (primarily serving duck clubs), and several state-managed wildlife refuges.

In this institutional context, multiple problems were addressed: (a) co-mingling of drainage and low-Se water supplies in local canals that supplied water to the wildlife refuges and (b) Se loads in excess of the assimilative capacity of the San Joaquin River (Quinn and Karkoski 1998). The initial action was an agreement to use a 38.6-km (24-mi) section of the old concrete-lined San Luis Drain as a dedicated bypass for agricultural drainage, eliminating the co-mingling of water and providing for a more accurate accounting of salt and Se loads. Once drainage had passed the wildlife refuges, it was released into Mud Slough and from there to the San Joaquin River. This, in turn, provided data for management of Se in drainage. Just as in other recent work in the Grasslands area, the new monitoring site in the San Luis Drain provided a basis for real-time monitoring and drainage management. The second element of the program was to establish a Regional Drainage Entity to develop and implement a drainage management plan, with an oversight committee to review operations and regulatory compliance. With authority to set fees and provide fee-based incentives to reduce Se loads in drainage water, the Oversight Committee was generally able to reduce Se loads, although exceedances may still occur during unusually high precipitation and runoff events outside of the range of controllable runoff. Sediment and biological monitoring programs for the San Luis Drain and Mud Slough were then established.

These drainage programs were combined with irrigation management programs, including a simple system for monitoring the depth of groundwater on a real-time basis, as a means for determining when to pump drainage water from the tile drain system. A tiered pricing for water was added to the management plan, with irrigation water equal to average evapotranspiration priced at a base rate, and water in excess of this standard priced at increased cost. This pricing program provides an economic incentive for growers to closely monitor water use and groundwater levels.

13.6 Conclusions

The Grassland Bypass and associated management programs is an example of the advantages and benefits of cooperative planning with multiple biological and agricultural objectives, organized at a regional scale, with active real-time monitoring and economic incentives to manage irrigation and drainage in a manner that avoids the need for regulatory action.

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Chapter 14
Regional Economics and Management in Closed Drainage Basins

Keith C. Knapp, Kurt Schwabe, and Kenneth A. Baerenklau

14.1 Introduction

This chapter develops a formal analysis of salinity and drainage management in arid and semi-arid regions. We consider a region with surface water imports but subject to a saline, high water table. As described in earlier chapters, irrigation water is applied in excess of plant requirements for salt leaching and also to overcome infiltration non-uniformity. As deep percolation flows add to the water table over time, they can encroach upon the root zone, resulting in yield losses via reduced aeration and/or salinization. Drainage systems and export can control water tables; however, environmental restrictions currently eliminate this possibility for much of the San Joaquin Valley (SJV).

One strategy for drainage management is to employ source control measures, such as changing crop mix, improving irrigation efficiency toward more capital-intensive, uniform infiltration systems, and varying irrigation timing and applied water to meet plant needs more closely. This strategy has undergone considerable analysis and some adoption (Knapp 1999). Growers can also reuse drainage water on salt-tolerant crops to provide both drainage disposal and water conservation services. This second strategy has received considerable attention in the scientific literature; however, there has been relatively little economic analysis [Knapp (1999) provides a literature review; Kan et al. (2002) is more recent work]. Additional strategies include evaporation ponds and land retirement. Evaporation ponds, which are common worldwide wherever restrictions exist on disposal into natural sinks, serve as an in-region disposal option (Johnston et al. 1997). Land retirement, alternatively, reduces the total amount of drainage by taking land, preferably with poor quality soils, out of production.

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_14,

The efficient solution to the salinity and drainage problem likely will consist of some combination of options. Unfortunately, integrated analyses of more than one option are relatively scarce. Knapp et al. (1986) is an early study considering source control (applied water reductions), drainage water reuse, and evaporation ponds but does not consider alternate irrigation systems. Hatchett et al. (1991) provide a policy simulation model with source control and reuse but not evaporation ponds. Posnikoff and Knapp (1996) evaluate source control, reuse with agroforestry, and ponds. Schwabe et al. (2006) evaluate an integrated system consisting of various source controls (deficit irrigation, changes in irrigation efficiency and crop mix) and drainage disposal (reuse, evaporation ponds). Wichelns (2005) provides a cost analysis of a sequential reuse system with solar ponds.

Regional economic management is addressed here with a sequence of three models, each of which couples an agricultural production economics model with a hydrologic model. The first model evaluates integrated drainage water management in down slope areas with a saline, high water table. The analysis finds that a high level of agricultural productivity can be maintained—principally via drain water reuse—without external drainage services. However, this model does not account for the fact that upslope deep percolation flows affect down slope areas. Consequently, we extend the analysis to a regional transect with declining land surface elevations and find that relatively minor upslope source control is efficient, due to discounting and finite hydraulic conductivities. A second limitation of the first model is an exogenous aquifer salt concentration. Therefore, the third analysis extends the regional management model to include endogenous water table elevations and salt concentrations.

Overall, the results support the idea that highly productive agricultural production operations can be maintained in closed drainage basins for a considerable period of time with purely in-region solutions. Furthermore, the analysis does not support widespread land retirement as an economically efficient option. These conclusions are subject to current prices, regulatory climate, and environmental conditions remaining at or at least not worse than, current conditions. However, sensitivity analysis does find that these conclusions are robust over a range of conditions.

14.2 Integrated Management with Saline, High Water Tables

This section develops the base model. It assumes a homogenous region with surface water imports overlaying a shallow, saline water table and with no external drainage facilities. This is a static (single time period) model for maximizing regional agricultural profits while maintaining regional hydrologic balance. The analysis follows Schwabe et al. (2006) which, in turn, builds on earlier contributions and represents conditions for drainage-impacted lands on the Westside SJV.

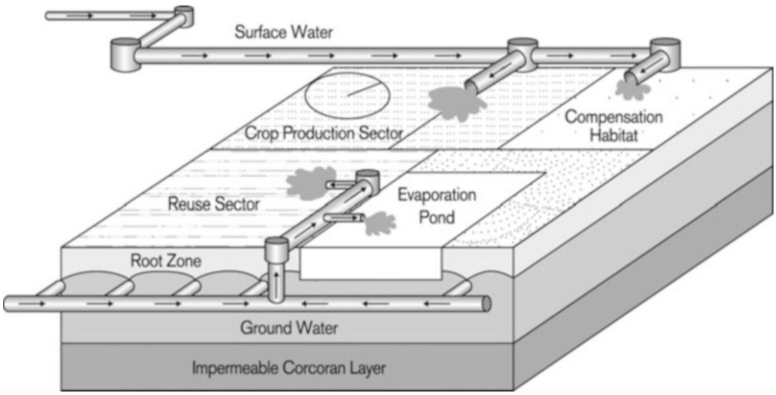

Fig. 14.1 Regional agricultural production with a shallow water table and no external drainage facilities

Figure 14.1 provides a schematic with three distinct sectors: freshwater crop production, reuse crop production from the saline groundwater source, and drainage disposal via evaporation ponds. Decision variables within the first two sectors are areas devoted to crop and irrigation system type and applied water rates, while evaporation pond area is the decision variable in the third sector. These variables are chosen to maximize basin annual net benefits subject to various constraints, including regional land area and a hydrologic balance constraint to stabilize the water table.

14.2.1 Mathematical Base Model and Data

Regional net benefits are given by:

$$\pi = \sum_{i=1}^{2} \sum_{j=1}^{n_{ic}} \sum_{k=1}^{n_s} \pi_{ijk} x_{ijk} - \psi_1\left(x_1^T\right) - \psi_2\left(x_2^T\right) - \eta_3 x_3 \tag{14.1}$$

where $\pi_{ijk} = \left(p_j^c y_{ijk} - \eta_{jk} - p_i^w w_{ijk}\right)$, w_{ijk} is applied water depth, x_{ijk} is land area devoted to crop production, and y_{ijk} is crop yield; and where i represents freshwater and reuse crop production sectors, j represents crop type, and k represents irrigation strategy. Applied water in the freshwater crop production sector ($w_{1\mathrm{jk}}$) comes from a surface water source, while water in the reuse sector ($w_{2\mathrm{jk}}$) comes from a saline groundwater source. The variable x_3 is area devoted to evaporation ponds. Parameters are p_j^c for the price of crop j, p_1^w and p_2^w for fresh and saline water prices, respectively, η_{jk} for nonwater production costs, and η_3 for evaporation pond costs. Although the analysis is for the region, all results are reported on a per-unit area (1-acre) basis.

The expression $\psi_i\left(x_i^T\right)$, where $x_i^T = \sum_{j=1}^{n_{ic}} \sum_{k=1}^{n_s} x_{ijk}$, reflects other costs associated with freshwater and reuse crop production, including additional production costs, land quality effects, or risk and uncertainty. Following Howitt (1995a, b), these costs are assumed to be quadratic in acreage. Crop yield and deep percolation flows for freshwater crops (d_{1jk}) and reuse crops (d_{2jk}) are defined as nonlinear functions of applied water depths with $y_{ijk} = f_{ijk}(w_{ijk})$ and $d_{ijk} = g_{ijk}(w_{ijk})$. While yield y_i and deep percolation d_i are functions of variable water applications, salt concentrations for both surface water imports and groundwater (drainage reuse) are exogenous. (Endogenous aquifer salt concentrations are considered in a later section).

The land area constraint is:

$$\sum_{i=1}^{2} \sum_{j=1}^{n_{ic}} \sum_{k=1}^{n_s} x_{ijk} + x_3 \le 1 \tag{14.2}$$

where a unit regional area is assumed for convenience. This constraint ensures that crop production activities do not exceed land area available for production. The hydrologic balance constraint is

$$\sum_{j=1}^{n_{1c}} \sum_{k=1}^{n_s} d_{1jk} x_{1jk} \le \sum_{j=1}^{n_{2c}} \sum_{k=1}^{n_s} (w_{2jk} - d_{2jk}) x_{2jk} + e_3 x_3 \tag{14.3}$$

where $d_{ijk} = w_{ijk} - et_{ijk}$ are deep percolation flows below the root zone, et_{ijk} is the evapotranspiration for sector i, crop j, and irrigation strategy k. The hydrologic balance constraint implies that deep percolation flows generated in the freshwater sector must be less than net disposals via reuse or evaporation, represented by e_3, in the pond. Equation (14.3) ensures the water table level is maintained at a level conducive to crop production.

14.2.2 Theoretical Analysis

Theoretical analysis with a simplified version of the above framework is reported in Schwabe et al. (2006). Efficient land use, irrigation strategies, and applied water rates depend on the opportunity cost of reuse and evaporation ponds. For example, the efficient amount of applied water depends not only on the water price, but also on the opportunity cost of drainage disposal. When a profitable reuse crop is available, or pond construction and operating costs are low, then these opportunity costs can be low (and potentially negative), thus reducing the need for intensive source control.

In the model, deep percolation flows must be reused or disposed in an evaporation pond. Typically, one expects that reuse will have lower evapotranspiration (ET) rates than an evaporation pond; yet, reuse may not only cost less but also generate positive net returns. The analysis shows that whether drainage water reuse or evaporation pond disposal is the method of choice depends, in part, on the opportunity cost of land: *ceteris paribus*, the higher the opportunity cost of land in agricultural production, the more attractive ponds become, given the assumption of uniform land quality. The efficient drainage option is thus an empirical question.

Sensitivity analysis was conducted for several parameters, including crop and water prices and aquifer salt concentration. An interesting finding is that the presence of general equilibrium effects can lead to counterintuitive results relative to partial equilibrium analyses. As an example, consider an increase in crop price. Normally, this would lead to increased water usage. However, in the drainage-constrained setting here, the effect is ambiguous: the increased crop price induces higher water use, but increases land opportunity costs and hence drainage costs, which dampens applied water rates. Overall, hydrologic balance at the regional scale implies a strong level of interconnectedness and nonseparability among the myriad activities, which greatly complicates economic efficiency calculations beyond those in a partial equilibrium framework, and also can lead to counterintuitive results.

14.2.3 Empirical Framework

The model is applied to conditions typical of Westland Water District (WWD). There are six crops varying in profit and salt tolerance (cotton, tomatoes, lettuce, wheat, alfalfa, and Bermuda grass). These crops historically account for more than 80 % of the region's cropped acreage. There are also six irrigation systems (furrow on a ½ mile run, furrow on a ¼ mile run, low energy pressure application, sprinkler, linear move, and drip) which vary in terms of capital and pressurization costs and water application uniformity.

Crop prices are derived from crop reports by the County Agricultural Commissioner, while surface water costs are estimated from water prices in the WWD. Nonwater production costs include planting, land preparation, weed cultivation, fertilizer, and tile and drainage systems costs. Crop production costs depend on crop type and irrigation system, while harvest costs include per acre (hectare) and yield-related components. Constraints are imposed to maintain acreages of individual crops within historical ranges observed during the 1990s. Finally, annual pond construction and maintenance costs are combined with compensating habitats to capture the expenditures associated with using evaporation ponds as a disposal alternative.

The crop-water production functions are developed for each crop-irrigation system based on plant growth models in Letey et al. (1985), and ET functional forms from van Genuchten and Hoffman (1984) and van Genuchten (1987). The functions express the relationships among yield, ET, salinity, and applied water (for complete description, see Kan et al. 2002). As salinity increases for a given level of applied water, yield deficits occur. Surface water and groundwater salt concentrations reflect actual salt concentrations in the region (Tanji and Karajeh 1993). Finally, the constrained maximization problem is solved using a nonlinear optimization procedure from the GAMS/CONOPT solver system.

14.2.4 Integrated Management

The first column in Table 14.1 describes a case in which there is unlimited external drainage. This provides a baseline intended to replicate historical conditions when growers either had access to external facilities or did not restrict emissions that led to water table buildup. All acreage is devoted solely to freshwater production subject to imposed fallow conditions at historical levels. Deep percolation flows are a little more than 1 ft year^{-1} (30.48 cm year^{-1}), consistent with observed empirical evidence, and there are substantial levels of agricultural net benefits defined here as returns to land and management.

The second column in Table 14.1 illustrates what happens if zero external drainage restrictions are imposed and only evaporation ponds are allowed. As shown, there is substantial source control, consisting of some acreage reduction and minimal crop shifting, but a significant shift to capital-intensive irrigation systems. Overall, these responses lead to a 37 % reduction in applied water, and deep percolation flows fall by approximately 15 % of the original amount. While the evaporation pond needed for hydrologic balance is small (6 %), the reduction in net benefits is quite large (34 %). This result is due to decreased acreage, more expensive irrigation systems, and possible yield reductions. Growers would be severely impacted in this scenario without outside assistance.

The third column in Table 14.1 allows for drainage water reuse. Here, substantial freshwater acreage shifts into reuse production; however, evaporation ponds are not used and the same total area of cropland exists, as in the unconstrained case. Interestingly, irrigation systems revert to the traditional systems of the unconstrained case, and applied water and deep percolation flows are also similar to the baseline case results. Most of the reuse production consists of cotton, which is both profitable and salt tolerant. Crop rotation constraints may be limiting the complete use of cotton for reuse. Most importantly, it can be seen that a high level of agricultural net returns are maintained—only 5 % less than the original level. Also reported is a drainage shadow value of \$19 acre-foot^{-1} (\$154 ha-m^{-1}) in the reuse case. This represents a value that—if imposed on grower emissions—would lead to the efficient solution. The same shadow value in the evaporation pond case is substantially higher. This value could also be used to determine efficient field-level decisions in a partial equilibrium analysis.

Schwabe et al. (2006) conduct a number of sensitivity analyses to test the robustness of these results with respect to a range of parameter values. Because the main solution depends on cotton reuse, several alternate cotton prices were considered. While a decrease in cotton price unambiguously lowers profits, it does so for both

Table 14.1 Drainage measurement under alternative drainage scenarios

	A	B	C
	External drainage	Evaporation ponds only ($x_{2jk} = 0$)	Reuse allowed ($x_{2jk}, x_3 \geq 0$)
Fresh crop production			
Area (acres of x_1^T)	0.83	0.75	0.51
Cotton (% of x_1^T)	58	53	37
Tomato (% of x_1^T)	24	27	39
Wheat (% of x_1^T)	4	4	–
Lettuce (% of x_1^T)	6	7	10
Alfalfa (% of x_1^T)	8	9	14
Water (acre – ft/year)			
Water use	3.28	2.05	3.18
Deep percolation	1.23	0.19	1.21
Irrigation systems			
FUR2 (% of x_1^T)	94	–	90
FUR4 (% of x_1^T)	–	13	–
Linear (% of x_1^T)	–	80	–
Drip (% of x_1^T)	6	7	10
Reuse production			
Area (acres of x_1^T)	–	–	0.32
Cotton (acres of x_1^T)	–	–	91
Wheat (acres of x_1^T)	–	–	9
Water (acre – ft/yr)			
Cotton (% of x_1^T)	–	–	3.85
Deep percolation	–	–	1.89
Irrigation systems			
FUR4 (% of x_1^T)	–	–	100
Land disposal (acres)			
Evaporation pond (x_3)	–	0.06	–
Welfare measures			
Net benefits ($) ~ π	$311	$206	$295
Drainage shadow value ~ λ_d ($/ac-ft)	–	$305	$19

Table results are from Schwabe et al. (2006) and are reported per unit of regional area (1 acre). The variable represents total area devoted to crop production in sector i. Column A External drainage refers to the case where there is no hydrological balance constraint

Water variables are average depths over the dropped areas per acre in each respective sector. FUR2 and FUR4 are, respectively, irrigation with furrow ½ mile and ¼ mile runs. Land areas and social net benefits are per regional acre with 17 % of land devoted to trees and fallowing

the unconstrained and constrained drainage cases. Even with a 23 % decrease in cotton prices, there is only a relatively small decrease in net benefits, and cotton reuse remains the preferred management strategy. Alternate aquifer salt concentrations, ranging from the original 10 dS m^{-1} concentration up to 20 dS m^{-1}, also were evaluated. Higher concentrations cause a gradual shift from reuse to source control to maintain hydrologic balance. Agricultural net benefits decline significantly over this range, while drainage shadow prices increase substantially, as would be expected. Still, land retirement and evaporation ponds are not observed over this range of aquifer salt concentrations.

14.2.5 Conclusions

The empirical results in Schwabe et al. (2006) suggest that agricultural production can be sustained over intermediate time horizons utilizing in-valley management options. While some source control is exhibited and while it can be anticipated that regions with low-quality land might use evaporation ponds, the primary mechanism for drainage management under the empirical conditions here is drainage water reuse. Furthermore, agricultural profits in the analysis are reduced by only 5 % relative to the unconstrained drainage case. This analysis assumes an exogenous level of lateral inflows from upslope areas and a constant aquifer salt concentration through time. These assumptions are relaxed in the next two sections, although it will be seen that the essence of the findings in this section are largely robust to these assumptions.

14.3 Dynamic Water Table Management, Up Slope Emissions, and Lateral Flows

The regional model in the previous section identifies efficient management strategies in a closed drainage basin. It suggests that a high level of agricultural productivity can be maintained over policy-relevant horizons through a combination of source control, reuse, and disposal (evaporation pond) strategies. The focus of the model is a closed drainage basin with exogenous lateral inflows. However, in many areas, including the Westside SJV, there can be upslope areas that generate drainage flows impacting down slope areas. Thus, an additional management strategy not considered in the preceding closed-basin analysis is reduction of lateral inflows to drainage-impacted regions.

This section addresses dynamic water table management. In particular, we are interested in (1) how water tables evolve in drainage-limited basins, (2) efficient management strategies that prevent or delay water table problems, and (3) the extent to which upslope source control is economically efficient. The model in this section is general and includes both spatial and dynamic aspects. Lumped parameter modeling of the evolution of a low-quality water table over time with uniform land quality and elevation is summarized in Knapp (1999). The main insight from the studies summarized there is that economic efficiency tends to restrain water table buildup via reduced net deep percolation flows relative to common property usage.

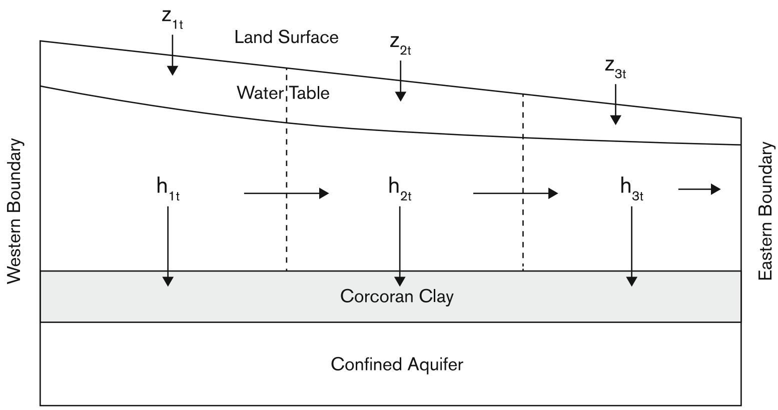

Fig. 14.2 Side view of the agricultural production/groundwater aquifer system on the Westside of San Joaquin Valley, California looking towards the north. The land generally slopes down from west to east. Vertical scale is exaggerated

14.3.1 Model

The model and analysis are based on Knapp (1996). As depicted in Fig. 14.2, we consider a transect generally aligned with the prevailing groundwater flow in a west–east direction and with the land surface sloping downward from west to east. The region is divided into a discrete number of cells to capture spatial variability. Net deep percolation flows are generated in each cell, and hydraulic heads in each cell evolve over time in response to net deep percolation flows in that cell, lateral flows from adjoining cells, and vertical flows through the Corcoran clay layer. The model aquifer is located along a transect of the area studied by Belitz et al. (1992). The transect is 19 miles (30.6 km) long and divided into 12 cells, with cell 1 being the upslope (western) cell, and cell 12 the down slope (eastern) cell. The land surface is assumed to slope uniformly down from west to east.

Cell net benefits are defined as $\pi(z_{it})$ where z_{it} denotes net deep percolation flows from cell i, $i = 1, \ldots n$, and year t, $t = 1, \ldots T$. The net benefit functions give social net benefits (profits) from agricultural production in each cell as a function of net deep percolation flows in that cell when all management strategies have been optimized subject to the various constraints. A programming model similar to Eqs. 14.1, 14.2 and 14.3 was used to estimate net benefit functions. Total cropping area is variable with 2/3 of the acreage devoted to cotton and 1/3 to tomatoes. Both the irrigation system type and applied water depths for each crop are also variable. Water can be extracted from the aquifer and disposed of in an evaporation pond. Irrigation system types, water quantity and land area devoted to crop production and the evaporation pond are then optimized subject to a constraint on net deep percolation flows and total land area.

In the estimated function, regional net benefits increase with net deep percolation flows, as would be expected. Net deep percolation flows are also subject to lower and upper bounds for cell-level net deep percolation, where the lower bound is negative, indicating all land devoted to the evaporation pond, and the upper bound is the profit-maximizing level of deep percolation absent a drainage constraint.

The water table height evolves over time in response to the existing hydraulic gradient and net deep percolation flows. Mathematically, this is written as

$$h_{t+1} = f(h_t, z_t) \tag{14.4}$$

where $h_t = (h_{1t}, \ldots h_{nt})$ is the vector of hydraulic heads at the beginning of time period t, and $z_t = (z_{1t}, \ldots z_{nt})$ is the vector of net deep percolation flows. This is the equation of motion for the dynamic system. Implicit within this equation system are lateral flows between cells, lateral flows into and out of the system, and vertical flows through the confining clay layer on the bottom of the aquifer. Hydraulic heads are subject to the constraint that $\underline{h}_i \leq h_{it} \leq \bar{h}_i$ where $\underline{h}_i$ and $\bar{h}_i$ are lower and upper bounds for hydraulic heads in each cell. The lower bound is determined by the aquifer lower boundary, while the upper bound for each cell is the maximum water table height, such that root zone integrity is maintained.

The equation of motion (Eq. 14.4) is specified using a finite-difference model. Lateral flows between cells and vertical flow through the clay layer are calculated according to Darcy's Law, and future hydraulic heads are computed via mass balance. Maximum allowable hydraulic heads are calculated as the elevation of the land surface minus typical depths to a drainage system, while minimum hydraulic heads are determined by the lower aquifer boundary. No-flow boundaries are assumed for both ends of the aquifer. This is realistic for the western boundary, but not necessarily for the eastern boundary; however, the latter is an overly conservative assumption for our purposes as it implies increased costs for the system. The various parameter values relating to aquifer geometry and hydrologic characteristics were generally computed from data in Belitz et al. (1992). The finite difference model is solved using monthly time steps.

14.3.2 Common Property

This model accounts for grower behavior at various points along the slope of the drainage basin. Under common property usage of the aquifer, producers make decisions to maximize their own profits without regard for impacts on others. With many users, it is reasonable to suppose that each also ignores the dynamic consequences of current-year decisions on future years (Provencher and Burt 1993). We therefore suppose that, in any given year, each cell chooses management strategies to maximize profits in that cell subject to available aquifer storage capacity in that cell. The latter is calculated as storage capacity at the beginning of the year adjusted according to expected lateral flows in that year.

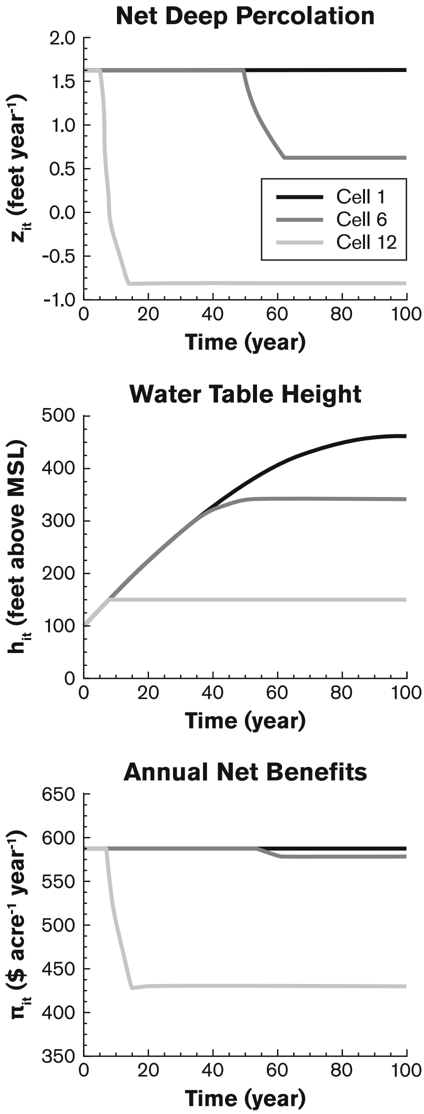

Fig. 14.3 Common property usage of the regional production/aquifer system for selected cells starting from $h_0 = 100$ ft mean sea level, net deep percolation (z_{it}), hydraulic head (h_{it}), and annual net benefit (π_{it})

Figure 14.3 illustrates common property dynamics starting from the assumed initial water table elevation (this is artificially low in order to illustrate the qualitative dynamics of the system). Initially, emissions in each cell start out at the (unconstrained) profit-maximization level of 1.5 ft year^{-1} (0.5 m year^{-1}). The water table rises uniformly over time until it reaches root zone levels in the down slope cell. After this point, emissions are successively reduced, beginning first with the down slope cells (i.e., cell 12), as the water table rises further over time. The system eventually converges to a steady-state with the steady-state being reached first in the down slope cells. Figure 14.3 suggests some overshooting in terms of net deep percolation levels, but this is relatively slight.

From Fig. 14.3, it can also be seen that net returns in all cells are positive over both the transition to the steady-state as well as the steady-state itself. Thus, agricultural production will remain in existence in all cells under these conditions. Also to be noted, however, is that there is a considerable disparity in returns depending on location. Some current landowners could be forced out of business if land was purchased at high prices. In the long-run, however, land rents and prices would adjust downward, reflecting lower net returns, and production would be maintained in all cells.

14.3.3 Economic Efficiency

Economic efficiency is defined as maximum present value of regional net benefits. Consideration of regional net benefits ensures taking into account all components of the system to the extent possible; present value ensures that investment in the resource is achieving the same rate of return as other investments in the economy. Under common property usage, emissions by one grower impose external costs on others. Since these external costs are not considered by profit-maximizing agents, common property usage of the aquifer is inefficient.

The present value of regional net benefits is

$$\sum_{t=1}^{\infty}\sum_{i=1}^{n}\alpha^{t}\pi_{it} \tag{14.5}$$

where α is the discount factor, T is the time horizon, and n is the number of cells. The objective is to choose annual net deep percolation levels to maximize the present value of social net benefits for the region (Eq. 14.5) subject to the equation of motion (Eq. 14.4), constraints on hydraulic head, and bounds on the net deep percolation levels. This problem is solved numerically as a running horizon problem using nonlinear optimization techniques.

Figure 14.4 illustrates aquifer dynamics under optimal management, beginning from an initial water table of 100 ft above mean sea level (MSL), which is artificially low in order to illustrate the general qualitative dynamics. As can be seen, water table heights are increasing over time. The system eventually approaches a steady-state, with the time to steady-state dependent on location. Generally, the down slope cells reach steady-state first and the upslope cells reach it later. Water table heights in the (approximate) optimal steady-state slope downward from west to east in response to reduced emissions in the eastern part of the transect. The maximum water table elevation is the land surface minus the root zone depth. In the steady-state, the water table elevation is below the maximal level for the four westernmost (upslope) cells and at the maximal level for the remaining cells. The results in

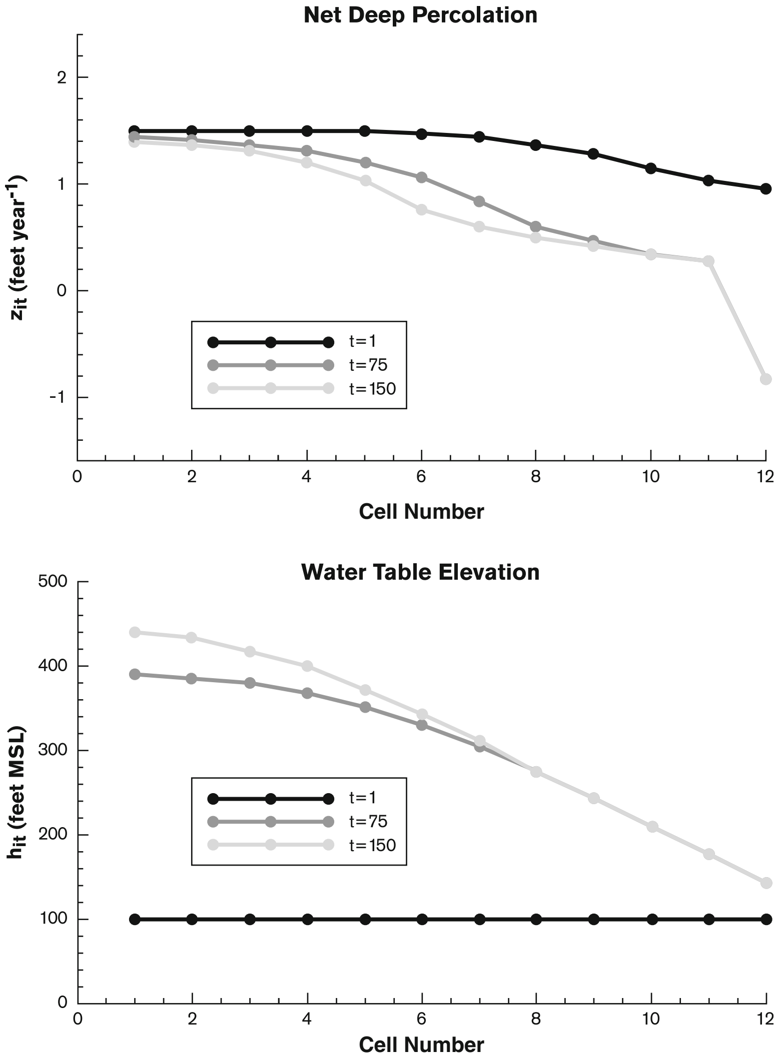

Fig. 14.4 Optimal management of the regional production/aquifer system for selected years starting from $h_{i0} = 100$ f. MSL. Net deep percolation (z_{it}) and hydraulic head (h_{it}).

Fig. 14.4 for net deep percolation flows exhibit the timing dimension of the lumped-parameter models; namely, under optimal management, emissions are progressively reduced as the pollutant stock increases.

In addition to the time dimension, management of this system is also subject to a significant spatial component. As Fig. 14.4 shows, net deep percolation in the optimal steady-state is declining as one moves down slope. What is most surprising is the magnitude of spatial variation. The western cells (1–5) retain relatively low levels of source control (high net emissions) ranging from 7–31 % reduction in deep

Table 14.2 Optimal management: Sensitivity of optimal net deep percolation in the first year to interest rates and hydraulic conductivity

			Deep percolation [z_{i1}*] (ft/year)		
r	K (ft/year)	K_c (ft/year)	Cell 1	Cell 6	Cell 12
0.02	6,300	0.189	1.49	1.38	0.19
	12,600		1.45	1.17	0.17
0.05	3,150	0.189	1.49	1.46	0.20
	6,300		1.49	1.45	0.17
	12,600		1.47	1.28	0.14
0.05	6,300	0.0	1.49	1.35	−0.18

Initial hydraulic head (h_{i1}) = 150 ft MSL
Parameters are (r) interest rate, (K) lateral hydraulic conductivity of the unconfined aquifer, (Kc) hydraulic conductivity of the Corcoran clay layer, and (z_{i1}*) optimal net deep percolation in cell i, year 1

percolation levels from common property levels with no drainage constraints. Intermediate cells (6–11) retain significant levels of net deep percolation (0.27 to 0.76 ft year^{-1} or 8.23–23.16 cm year^{-1}), while the down slope cell (cell 12) has negative net deep percolation. Although the down slope cell serves as a drain for the rest of the system, agricultural production remains viable in that cell with net returns of \$430 acre^{-1} (\$1,063 ha^{-1}) in the approximate steady-state. The magnitude of spatial variability in source control is nominally surprising since the transect length is relatively short (19 miles or 30.6 km), and the overall land slope (376 ft or 114 m over this distance) is relatively small and not noticeable upon visual inspection. This is also significant as previous analyses (SJVDP 1990) propose relatively uniform policy standards over areas of this size. The underlying motivation and explanation for this finding is explored next.

The observed spatial variability in efficient management appears to be due to a combination of land slope, finite aquifer transmissivity, and discounting. Because of the land slope, the elevated water table impacts the down slope cell first. In addition, the aquifer has finite transmissivity, implying that emissions from upslope areas will not fully affect down slope water table levels for some time. Combined with a positive discount rate, the present value of damages in the down slope cell from a unit of net deep percolation emissions in cell i progressively decrease as one moves upslope. This decline implies reduced source control for upslope generators in comparison to down slope generators. With regard to the land slope, a flat land surface would correspond to the basic stock pollutant model and net deep percolation levels would be constant across space. The remainder of the hypothesis—transmissivity and discounting—is tested in Table 14.2, which reports a sensitivity analysis with respect to aquifer transmissivity and the interest rate. The initial hydraulic head is a uniform 150 ft (45.7 m) above mean sea level (MSL) for all cells, and the effects on first-period net deep percolation levels are considered.

Decreasing the interest rate increases source control for the upslope and intermediate cells. A decrease in the interest rate places relatively more weight on the future and hence increases the costs associated with net deep percolation from these cells. An increase in lateral hydraulic conductivity increases source control in the upslope cells. This is as expected, since increasing hydraulic conductivity implies that damages from deep percolation in a given cell will be felt in the more immediate future. Net deep percolation in the down slope cells also falls with the hydraulic conductivity. Although net deep percolation is reduced in the upslope cells, this is counterbalanced by the increase in flows due to the increased hydraulic conductivity and hence, net deep percolation falls in the down slope cells. Overall, the conclusion that significant upslope source control is inefficient remains robust to significant changes in these parameters. Sensitivity analysis was also performed for the assumption of a constant head for the confined aquifer underlying the Corcoran clay, and for the no-flow eastern boundary assumption. In both instances, the overall qualitative finding of minimal upslope source control remained robust to the assumed boundary conditions.

14.3.4 Policy Analysis

Policy implications follow upon comparison of common property and efficiency. Here, the initial water table level is that reported in Belitz et al. (1992) for 1984: hydraulic head is approximately 230 ft (70 m) MSL for the first six cells, and then roughly follows the land slope minus the drain system depth (8 ft or 2.4 m) after that. The top diagram in Fig. 14.5 contrasts common property and optimal net deep percolation depths in the first year, given the initial water table level. As can be seen, optimal depths in the upslope four cells are approximately equal to common property levels. Optimal values are significantly less than common property levels for the intermediate cells 6–11. For the down slope cell 12, however, optimal net deep percolation levels are actually greater than common property levels; this follows from the upslope source control. The present value of regional net benefits over a 25-year horizon is \$8,124 acre^{-1} (\$20,075 ha^{-1}) under optimal management and \$8,081 acre^{-1} (\$19,969 ha^{-1}) under common property usage. Thus, the annualized social gains from managing the resource are \$3.09 acre^{-1} year^{-1} (\$7.64 ha^{-1} year^{-1}).

An important concept for dynamic efficiency and optimal regulation is marginal user cost (MUC). In this context, the MUC for a particular cell can be interpreted as the present-value of future damages in the basin from a 1-unit increase in deep percolation emissions in that cell in the current year. Mathematically, the MUC in cell i, year t is precisely defined as $\sum_{j=1}^{n} \lambda_{j,t+1} \frac{\partial f_j(h_t, z_t)}{\partial z_{it}}$ where $\lambda_{j,t+1}$ is the Lagrange multiplier associated with the hydraulic head equation of motion in cell j in the optimization model. The Lagrange multiplier values are recovered from the nonlinear programming solutions in the running horizon solution method. The derivative values are computed exactly with a recursive algorithm inside the finite-difference transport model.

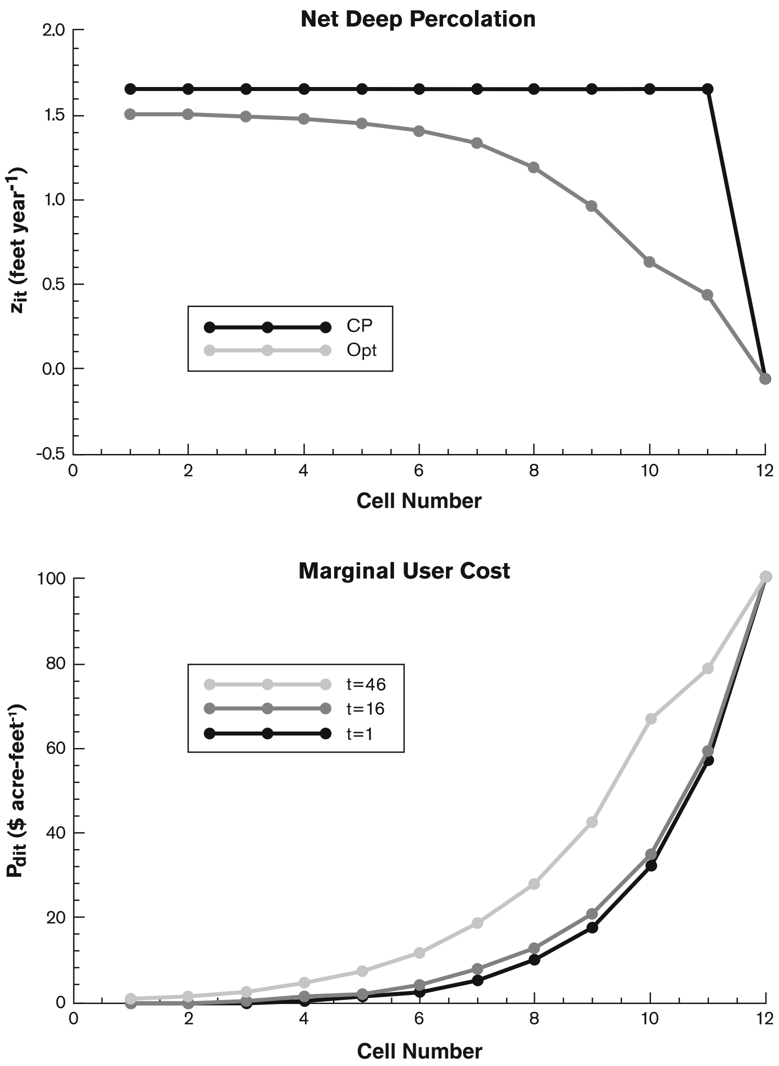

Fig. 14.5 Regulatory policies for selected years starting from the 1984 water table. (**a**) Net deep percolation in year 1 under common property (*CP*) and optimal management (*Opt*) (**b**) Marginal user cost of deep percolation by cell for selected year

The bottom diagram in Fig. 14.5 illustrates the MUC of deep percolation under optimal management. First-year MUC ranges from zero for the upslope cell to more than \$100 acre-foot^{-1} (\$810 hectare-m^{-1}) for the down slope cell. Marginal user costs are lower for the upslope cells due to the finite aquifer transmissivity and discounting, implying that damages from emissions in upslope cells are delayed into the future relative to down slope cells in which damages are felt more immediately. The MUC increases over time as the water table rises consistent with the lumped-parameter analysis. The implication for regulatory policy is that setting an emissions charge on net deep percolation flows according to these values would induce economic efficiency.

14.3.5 Conclusions

The main findings from this analysis can be summarized as follows:

- Net deep percolation flows are progressively reduced as the water table rises under economically efficient aquifer usage. This finding is consistent with the previous lumped-parameter analysis.
- Somewhat surprisingly, minimal support is provided for upslope source control. Even over long time horizons, emissions in the upper elevations remain at or near profit-maximizing (common property) levels.
- These results appear to be due to a combination of land slope, finite aquifer transmissivity, and discounting. The finite aquifer transmissivity implies a time delay between upslope emissions and down slope impacts. Combined with a positive discount rate, the present value of marginal damages is therefore lower for upslope areas than for the down lope areas, implying lower source control upslope than down slope. This result is robust to sensitivity analysis in the interest rate, hydraulic conductivity, and boundary conditions.
- Optimal emission charges increase as the water table rises and are higher down slope than upslope.

These topics need further investigation due to the potential significance of the results (inefficiency of upslope source control), model complexity, limitations of the analysis, and inherent difficulties in solving large-scale nonlinear optimization problems.

14.4 Endogenous Salt Concentration and Water Table Elevation

Assuming that the groundwater salt concentration is a fixed, exogenous parameter and that the water table elevation is held constant by the hydrologic balance constraint, regional management in Section 2 of this chapter (Integrated Management with Saline, High Water Tables) finds that saline groundwater reuse can provide an economical solution to the problem of drainage water management. However, as salts are imported into the region in surface water, the groundwater salt concentration would tend to increase through time, rather than remain constant; the water table elevation also may change. These observations suggest that, contrary to the assumptions in Sect. 14.2, changing aquifer characteristics may be economically relevant and may impact the viability of the reuse strategy.

Groundwater salinity economics has received relatively little attention in the literature. Cummings (1971) and Cummings and McFarland (1974) develop conceptual frameworks and interpret efficiency conditions. Dinar (1994) and Dinar and Xepapadeas (1998) investigate energy costs, surface water limits, and pumping taxes for agricultural producers overlying a common aquifer, while Zeitouni and Dinar (1997) model a two-aquifer system in which subsurface flows from a high-salinity aquifer degrade a low-salinity aquifer. Reinelt (2005) considers seawater intrusion due to drawdown from irrigated agriculture. Roseta-Palma (2002) provides a theoretical investigation of groundwater quantity and quality, but assumes a natural regenerative capacity for the aquifer and focuses on the steady-state, in contrast to this application in which salinity is a conservative pollutant and transition times can be very long. Zeitouni (1991) develops a very sophisticated theoretical model of groundwater quality.

This section analyzes dynamic reuse by extending the Schwabe et al. (2006) model described in Sect. 14.2 to include aquifer dynamics for water table elevation and aquifer salt concentration. These aquifer dynamics follow the Knapp and Baerenklau (2006) saline groundwater model, in which the equations of motion are based on mass balance calculations for water and salt flows, and define water table elevation and salt concentration evolution over time. Efficient dynamic management is investigated and, in particular, the results of Sect. 14.2 are found to be valid over reasonable policy-relevant time horizons (several decades). This result is due to the fact that annual deep percolation flows and salt loading to the aquifer are small relative to the aquifer size, leading to a relatively slow increase in aquifer salt concentration.

14.4.1 Mathematical Model and Data

Consider a region with a potentially high saline water table where both the water table elevation and the aquifer salt concentration evolve over time (Fig. 14.6). Net deep percolation flows are defined as deep percolation flows from agricultural production, less extractions from the aquifer. The region may undertake actions to affect net deep percolation flows, including source control, reuse, and disposal in evaporation ponds. The aquifer then evolves through time in response to net deep percolation flows. If the water table becomes exceedingly high, or if highly saline water is used for irrigation, damage to crop production may result. The economic objective is to maximize the present value of regional net benefits from agricultural production, subject to aquifer dynamics.

The present value of regional net benefits from agricultural production is

$$\pi = \sum_{t=1}^{\infty} \alpha^t \pi_t \tag{14.6}$$

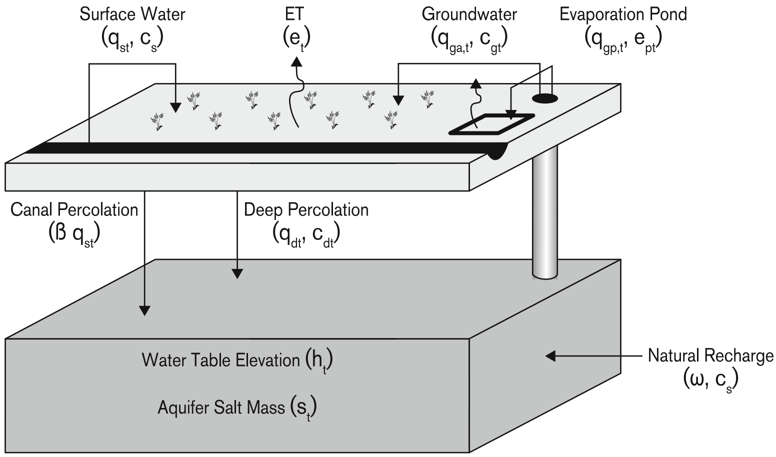

Fig. 14.6 Schematic diagram of an agricultural production/saline aquifer system

where α = the discount factor and annual net benefits are given by

$$\pi_t = \sum_{i=1}^{2}\sum_{j=1}^{n_{ic}}\sum_{k=1}^{n_s}\pi_{ijkt}x_{ijkt} - \psi_1\left(x_{1t}^T\right) - \psi_2\left(x_{2t}^T\right) - \eta_3 x_{3t} \tag{14.7}$$

where $i = \{1,2\}$ denotes freshwater and reuse crop production sectors, j denotes crop type, and k denotes irrigation system type. Crop yields, deep percolation flows, and the land area constraint remain as in Sect. 14.2 and also apply to each time period.

The hydrologic balance constraint in Sect. 14.2 is replaced by an equation of motion describing water table elevation h_t response to net deep percolation flows

$$h_{t+1} = h_t + \frac{1}{As^y}\left(\omega + \beta_s q_{st} + \sum_{i=1}^{2}\sum_{j=1}^{n_{ic}}\sum_{k=1}^{n_s} d_{ijt}x_{ijt} - q_{gt}\right) \tag{14.8}$$

where A is the regional aquifer area, s^y is the aquifer specific yield, ω is natural recharge, β_s is the surface water infiltration coefficient, q_{st} is total surface water imports, and q_{gt} is total groundwater extractions. This equation states that the water table elevation rises with natural recharge, percolation from surface water imports (canal losses), and irrigation; elevation falls with extractions for irrigation and disposal. The water table elevation is constrained by $\underline{h} \le h_t \le \bar{h}$, where $\underline{h}$ is determined by the lower confining layer and $\bar{h}$ is determined by the root zone depth. The lower bound limits groundwater extractions to the available supply and the upper bound limits net deep percolation flows to the maximum storage capacity that is feasible for crop production.

Given the long time horizon considered here, two assumptions are made to characterize the quality dimension of the aquifer: (a) uniform mixing in the aquifer, and (b) steady-state conditions in the root zone. The latter is supported by the relatively fast dynamics of soil salinity compared to groundwater salinity. Groundwater salt concentration is thus calculated as: $c_{gt} \equiv s_t/(As^y(h_t - \underline{h}))$, where s_t is the aquifer salt mass. The equation of motion for the salt mass, which together with the water table equation of motion implicitly defines salt concentration dynamics, is given by

$$s_{t+1} = s_t + c_s q_{st} + c_\omega \omega - c_{gt} e_p x_{3t} \tag{14.9}$$

where c_s and c_ω are the salt concentrations for surface water and natural recharge, respectively, and e_p is the pond evaporation rate. This equation states that salts are brought into the system through surface flows and natural recharge and are removed through disposal via evaporation ponds. Under steady-state root zone conditions, salt inflows from deep percolation due to irrigation with groundwater exactly cancel salt outflows due to groundwater extraction and thus neither appears in the equation.

14.4.2 Efficient Allocations Over Time

For economic efficiency, the objective is to choose annual freshwater imports and groundwater extractions, crop and irrigation system types, the total acreage for each combination of inputs, and the evaporation pond area to maximize the present value of regional net benefits from agricultural production. The equations of motion link the time periods together and specify how the aquifer characteristics evolve through time in response to the choice variable selections. The problem is solved for Westside SJV characteristics over an infinite horizon using dynamic programming (Bertsekas 1976).

The solution to this optimization problem can be represented by a set of time paths for the aquifer characteristics and the control variables. Figure 14.7 shows the optimal path for water table elevation. There is essentially no change in the water table elevation through time, which is consistent with observations under limited drainage conditions. The water table is managed effectively as a nuisance, rather than as a beneficial resource, with its elevation held steady just below the root zone. Notably, this implies the hydrologic balance constraint imposed on the mathematical model of Sect. 14.2 is satisfied by this optimal solution; in other words, invoking the hydrologic balance constraint as a necessary simplification to forego modeling the dynamics of the water table has no discernable effect on the model results.

Figure 14.7 also shows the optimal path for the aquifer salt concentration. Unlike the water table elevation, the salt concentration changes through time as salts build up in the aquifer, primarily due to deep percolation flows. This result differs from

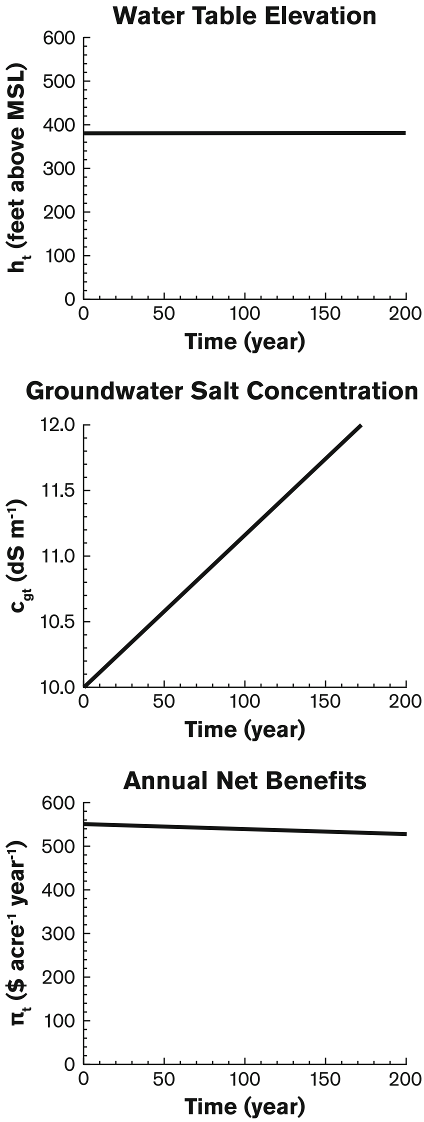

Fig. 14.7 Time-series for the aquifer variables and annual net benefits in the agricultural production/saline aquifer system

the corresponding assumption in Sect. 14.2, namely that the aquifer salt concentration remains constant through time. However, noting the scales in Fig. 14.7, the salt buildup occurs relatively slowly: it takes about 85 years for the salt concentration to increase by 10 %, from 10–11 dS m^{-1}. Furthermore, with a discount rate of 4 % annually, \$1 of annual net benefit received 85 years in the future has a present value of less than \$0.04. Therefore, it seems plausible that, in addition to the water table elevation, the salt concentration also can be assumed constant throughout the relevant planning time horizon; however, before drawing this conclusion, it is first necessary to examine the results for the control variables.

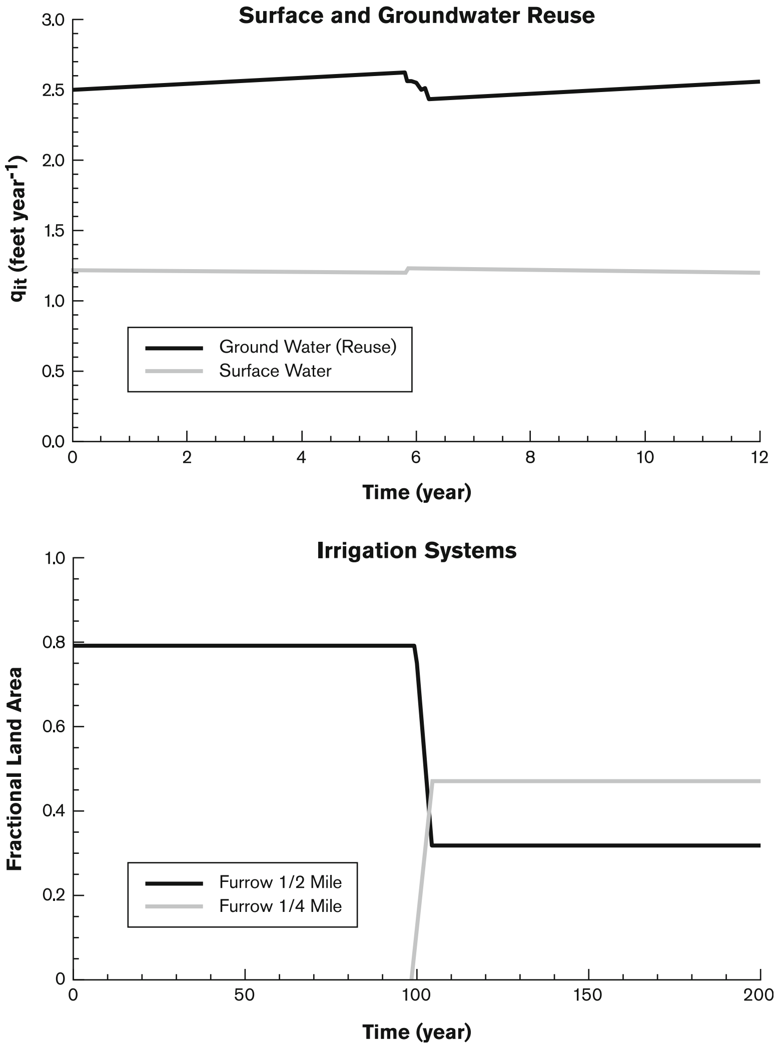

Fig. 14.8 Optimal management of the agricultural production/saline aquifer system over time. (**a**) Surface water and groundwater reuse, (**b**) irrigation systems

The top diagram in Fig. 14.8 shows the optimal time paths for total surface water imports and total groundwater extractions. Surface water imports decline gradually during the first 100 years, approximately, followed by a brief increase and then another gradual long-term decrease. Groundwater extractions follow the opposite path: two long-term periods of gradual increase interrupted by a brief decrease. Overall, each remains relatively constant through time and similar in magnitude to the optimal use levels for the Sect. 14.2 model. The brief interruptions in these otherwise monotonic trends correspond to a change in irrigation system technology.

This is illustrated in the bottom diagram of Fig. 14.8, which shows farmers optimally switching acreage from a ½ mile (805 m) run furrow system to a ¼ mile (403 m) run furrow system at this time. The use of other irrigation systems remains constant and relatively minimal over the horizon. Moreover, the predominant use of a furrow, ½ mile (805 m) run system during the first 100 years matches the optimal irrigation system choice from Sect. 14.2. Not shown are the optimal time paths for crop areas, as each is essentially constant and corresponds closely to the optimal areas derived in Sect. 14.2.

14.4.3 Conclusions

The findings in this section suggest that the results from Sect. 14.2 are quite robust with respect to the assumptions made about constancy of the water table elevation and the aquifer salt concentration. Allowing both of these to vary does not generate significantly different results over a reasonable time horizon. Primarily, this is due to the relatively slow evolution of the aquifer system as well as the effects of discounting. An additional insight provided by this multi-period analysis is that annual net benefits gradually decline through time as the aquifer salt concentration increases. Therefore, agricultural production is not economically "sustainable" in the long run, as might be interpreted from the steady-state analysis of Sect. 14.2. However, the optimal management strategy remains essentially unchanged when the simplifying assumptions about water table elevation and aquifer salt concentration are relaxed.

14.5 Qualitative Conclusions Regarding Closed Drainage Basins

The chapter addresses the economics of salinity and drainage management in closed drainage basins. The analysis is developed for average conditions (spatial and temporal) on the Westside SJV; thus, the conclusions would not apply equally to every field in every year. The conclusions are intended to hold, on average, and to be representative for the region as a whole. With this caveat, the primary qualitative conclusions are summarized below.

14.5.1 Reuse

Absent external drainage facilities, a variety of biophysical management strategies can be used to stabilize saline water tables. The results here point primarily to drainwater reuse as a mechanism for water table stabilization. Cotton is an ideal crop for reuse, as it is relatively salt tolerant and avoids potential health issues associated with drainage irrigation on food crops, but does depend on market conditions. Future research could profitably investigate the potential of drainwater reuse for bioenergy production.

14.5.2 Source Control and Evaporation Ponds

The analysis finds relatively little scope for source control beyond cost-effective improvements to traditional irrigation systems. There is relatively little crop-switching in the least-cost solutions, and high crop yields are maintained; thus, moisture-stressing also does not appear as a very significant strategy. Little scope was also found for evaporation ponds, although this conclusion is driven by the uniform land quality assumption and the consequent high land opportunity costs. If marginal lands are available, then evaporation ponds are potentially more desirable.

14.5.3 Land Retirement

The results support the proposition that widespread land retirement is not necessary for a sustainable system with profitable agriculture and maintained water table levels. This conclusion is strengthened by two factors not considered here: (a) land retirement could have adverse impacts on employment and agribusiness facilities in the area, implying transition and equity effects for local populations; and (b) land retirement could lead to adverse environmental impacts, if the saline water table rises close to the land surface in downslope areas. However, land retirement may still be justified for local "hot spots" with naturally high levels of selenium, for example.

14.5.4 Upslope Source Control

Upslope areas can contribute physically to drainage problems in downslope areas without themselves being affected by high water table conditions. Somewhat surprisingly, this analysis did not find upslope drainage management to be economically efficient. The reason for this is a combination of low hydraulic conductivities leading to long time-delays, and the presence of discounting to ensure investment efficiency in the economy as a whole.

14.5.5 Sustainability

The results suggest that agricultural production in the region is sustainable over intermediate (decadal) periods. High levels of productivity and profitability can be maintained over policy-relevant time horizons, but as salts are imported into the region, and if groundwater reuse just re-circulates salts in the system over reasonable time scales, it is physically certain that reuse cannot be relied upon forever because the water table salt concentration eventually will increase, absent external drainage. Thus, reuse should not be considered an indefinite solution. However, the results suggest that the groundwater salinization process is likely to occur very slowly on average and is not limiting on at least decadal time-scales. As an aside, these results also demonstrate that the salinity/drainage problem in the SJV is not due primarily to salt importation, as sometimes portrayed, but rather is an outcome of natural salinization due to soils of marine origin on the Westside.

14.5.6 Study Limitations

The research in this chapter has limitations that can affect the conclusions in specific situations. As mentioned, the analysis assumes uniformity with respect to regional land quality, water supply conditions, and operator characteristics, but does include field-level spatial variability and land elevation variability. This implies that in specific areas or years differing significantly from the average, one could arrive at different conclusions regarding evaporation ponds and land retirement, as noted above. While ideally one would try to expand the analysis here to include more heterogeneity, this will be challenging when trying to maintain detailed, verifiable, micro-level representations.

An alternate course is to design policy instruments to capture heterogeneity not specifically in the model. As an example, Schwabe et al. (2006) find a shadow value of \$19 acre-foot^{-1} (\$154 ha-m^{-1}) for deep percolation flows. Suppose that an emission charge/reuse subsidy based on this value were utilized by a water district. In that case, operators with beneficial conditions might practice more source control and/or reuse than the average, while operators subject to adverse conditions would practice less. Thus, the results for models with representative conditions can still provide a fundamental basis for policy design and implementation even though not all uncertainty and variability are accounted for, something which is impossible from a modeling perspective, in any case.

There are also major areas not addressed by this research needing future work. First, this research does not address out-of-valley disposal, an option that needs formal benefit-cost evaluation for comparison with in-valley solutions. However, for reasons already given, some level of in-valley management is likely efficient, even with out-of-valley disposal opportunities. Another major research area is environmental valuation of evaporation ponds, which are currently subject to stringent regulation. These regulations may be based implicitly on a zero-harm criterion and not necessarily on a balance between costs and degree of ecosystem impact. Also, at least in theory, compensating habitat requirements imply that wildlife could benefit potentially from a system consisting of both evaporation and solar ponds, along with additional set-aside acreage managed exclusively for wildlife.

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Chapter 15
California and Beyond: An International Perspective on the Sustainability of Irrigated Agriculture

Dennis Wichelns and James D. Oster

The challenge of achieving sustainable irrigation by preventing salts from accumulating in irrigated soils is not unique to California. Salinity and drainage problems have required attention in irrigated areas throughout the world for thousands of years. In some areas, farmers, regional associations, and public agencies have addressed the problems with moderate success. In many others, secondary salinization and waterlogging continue to reduce crop yields and farm incomes, with consequent, negative impacts on livelihoods and food security, particularly in developing countries.

The widespread and persistent occurrence of waterlogging and salinity suggests that the causes and complications associated with irrigation and drainage are fundamental, and they apply globally, across a wide range of geographic and cultural conditions. In most areas, the two primary underlying causes of waterlogging and salinity are inappropriate irrigation water management and delayed construction of an adequate drainage system. The causes and impacts are similar throughout the world, as are the spatial and temporal dimensions. In addition, potential solutions are constrained by similar physical, economic, and societal factors.

Problems associated with toxic elements in drainage water, including selenium, are not unique to the San Joaquin Valley or to arid areas within the United States. Selenium and other toxic elements, such as arsenic, are found in shallow groundwater and drainage water in India, Bangladesh, Chile and Argentina, and many other nations throughout the world. The common theme in the occurrence of selenium and arsenic in these waters appears to be either: (1) the parent materials

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A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2_15,

of soils in the region are shales that were deposited under oceanic conditions, as is the case on the west side of the San Joaquin Valley, or (2) the groundwater used for irrigation has been influenced by ocean sediments. As a result, high concentrations of selenium are found in soils and shallow groundwater in the San Joaquin Valley and in groundwater along the coastal range that defines the Valley's western edge.

Based on the case studies discussed in this chapter, which span three continents (Asia, Australia, and North America), it can be argued that salinity and drainage problems and the associated mobilization of toxic chemicals in water arise in arid regions due to the multiple interactions of the following complex factors:

- Technical relationships involving soils, water, crops, and irrigation;
- Institutional issues, including conflicting time horizons, property rights, prices, and market structure; and
- Government intervention, including the failure to enact region-wide drainage plans in conjunction with the approval of irrigation projects

These three complex factors, in concert, contribute substantially to the genesis and persistence of salinity and drainage problems. Our perspective is consistent with Hillel's view (1991), who suggests that salinity and drainage issues arise and persist over time due to a combination of technical, agronomic, and management issues. Unexpected issues, such as the occurrence of selenium or other toxic chemicals or trace elements in agricultural drainage water, also may arise over time and complicate efforts to solve salinity and drainage problems.

The first factor we have identified, technical relationships, has been discussed and explored in detail in previous chapters. This chapter focuses on the second and third factors, institutional issues and government intervention. Following an analysis of both factors, we review three specific irrigation and drainage case studies in Pakistan, Turkey, and Australia, in which waterlogging and salinity are being addressed by national governments and regional irrigation and drainage organizations, with differing levels of success. We also analyze the lessons learned about the sustainability of irrigated agriculture in California's San Joaquin Valley, highlighting the institutional and governmental response to selenium contamination and the regulatory decisions implemented for environmental protection. Using both the international case studies and the ongoing work on the west side of the San Joaquin Valley in California, we describe the role of institutions and governments in creating salinity and drainage problems and in crafting approaches that generate sustainable solutions

15.1 Institutional Issues and Government Intervention

15.1.1 A Long-Term Perspective Is Necessary for Sustainable Irrigation Management

> If a market-guided system is to perform well over the long haul, it must be more than myopic. Someone – it could be the Department of the Interior, or the mining companies, or

> their major customers, or speculators – must always be taking the long view. They must somehow notice in advance that the resource economy is moving along a path that is bound to end in disequilibrium of some extreme kind. If they do notice it, and they take defensive actions, they will help steer the economy from the wrong path toward the right one. – Robert M. Solow (Nobel Laureate in Economics), speaking at the annual meeting of the American Economic Association, 1974 (Solow 1974).

Although Professor Solow's keynote address focused specifically on rapidly rising oil prices in the United States, caused largely by an unexpected reduction in supply due to sharply lower exports from oil-rich nations, his caveat about the need for a long-term perspective has unmitigated application to the twin problems of waterlogging and salinity in irrigated agriculture. Professor Solow described the tendency of market participants to focus too often on near-term costs and benefits, while giving too little attention to long-term implications of resource extraction and consumption. For irrigation management programs to be sustainable and to ensure long-term optimality of water and land use, "[S]omeone… must always be taking the long view," as Dr. Solow wisely warned.

Diametrically opposed to Dr. Solow's perspective is the fact that most farmers in arid regions throughout the world place great emphasis on the near-term costs and benefits of irrigation. Where successful crop production requires supplemental water, farmers are eager to obtain irrigation service from rivers, surface canals, or groundwater because it enables them to achieve crop yields that provide household food security directly or to generate income for purchasing supplies in local markets.

> For the farmers, irrigation is a need for today ("no water no crop") and salinity is a problem of tomorrow. (Ritzema et al. 2008)

Most farmers, whether poor, rural residents of developing countries, or much wealthier large-scale growers in industrialized countries, focus on the near-term costs and benefits of irrigation, while giving too little attention to the long-term problems of waterlogging and salinity, which have developed in many arid regions worldwide, becoming extensive and quite costly for farmers, their communities, and their countries, in some cases. Thus, the San Joaquin Valley story attracted considerable attention. The initial focus of farmers and public agencies on irrigation, while delaying consideration of a regional drainage system, is typical of the near-term focus observed in many irrigated areas which predictably shortchanges the sustainability of the water and land use program.

15.1.2 Sustainable Irrigation Requires Effective Management of Water and Salts

> Waterlogging in irrigation projects is an age-old problem dating back to civilizations in the Euphrates and Tigris regions in Mesopotamia. These civilizations had serious waterlogging problems from which, even after centuries, mankind has learnt no major lessons. (Bowonder et al. 1987).

Is there widespread evidence to support the sweeping indictment by Bowonder et al. (1987) that "mankind has learnt no major lessons" since the waterlogging failures that occurred in Mesopotamia more than 4,000 years ago? Many reports describing the twin problems of waterlogging and salinity in modern times refer to the epic and long-lasting salinity-induced decline of early settlements in Mesopotamia, noting that farmers around the world have been working for millennia to prevent or adapt to these problems, particularly in arid regions, but with limited success. In Mesopotamia, during the period from 2400 to 1700 B.C., in a region now known as southern Iraq, salinity wrought its devastating effects on their ancient society. Water was diverted from the Tigris and Euphrates rivers to irrigate fertile lands. Excess quantities of irrigation water were applied; evaporation rates were high; and water tables were shallow, and when these features of the irrigation program were combined with the flat topography of the Mesopotamia plain, they led to water table rise, waterlogging, and the ultimate salination of the soil by capillary rise. Crop yields declined, Fields abandoned, and cultivation shifted to more salt-tolerant crops as the soil salinity increase. Despite the huge effort in building intricate surface and subsurface drainage networks, the southern part of the alluvial plain in Mesopotamia never recovered. Salinity had contributed to transforming fertile soils into desert conditions (Jacobsen and Adams 1958).

Many of the world's finest scientists, technicians, and public officials have engaged in long-term efforts to understand waterlogging and salinity, and to devise new strategies to prevent or adapt to the problems in selected areas. Substantial financial commitments have been made for teaching, research, and outreach efforts to develop measures that might prevent the future occurrence of waterlogging and salinity, as is evident in California's Central Valley and in the modern case studies in Pakistan, Australia, and Turkey described in this chapter. Yet, problems continue to arise, even in regions with advanced technology and institutions.

To place waterlogging and salinity in proper historical perspective, in the millennia that have elapsed since these noxious twins were first linked to excess irrigation and inadequate drainage, some of the world's most challenging medical problems have been overcome and many notable frontiers have been crossed. For example, humans have visited the moon, robots have explored the surface of Mars, and anyone with access to the Internet can explore the universe. Smallpox has been eradicated, polio nearly so, and some forms of cancer have been treated in children and adults with success, but waterlogging and salinity continue to persist, reducing farm incomes and constraining rural livelihoods in many areas worldwide. Surely, technology cannot be the factor that limits man's ability to eradicate waterlogging and salinity.

15.1.3 Disconnect Between Individual Gains and External Impacts Off-Site Favors Buildup of Salinity and Waterlogging

If technology is not limiting a successful response to waterlogging and salinity in drainage, more fundamental institutional issues must be responsible for the persistence of these problems in both basic and technologically advanced irrigated agricultural settings.

While farmers benefit directly from the use of irrigation water, any negative impacts of their individual activities on regional waterlogging and salinity problems are difficult to identify and cumulative over time. Each farmer's irrigation management practices contribute to the volume of water percolating through the root zone or flowing through surface drainage channels, eventually placing pressure on regional shallow aquifers and drainage collection basins. It is difficult to quantify the precise impacts attributable to individual farmers and, hence, farmers often are not held responsible for the water or salt they discharge, particularly in the early years of an irrigation project. This disconnect between the farm-level benefits of irrigation and the off-farm costs is exacerbated by the typical time-lag between initiation of irrigation and drainage and the accumulation of salts to a level that may be considered a "problem". The time delay can perpetuate until a regional waterlogging and salinity problem becomes severe. At that time, public agencies often are called upon to devise solutions (Datta and deJong 2002).

Economists often describe this disconnect between individual gains and external impacts within the context of externalities, market failure, or missing markets. When individuals, such as farmers, lack property rights to key resources, such as water, or do not face penalties for harm imposed on others, negative impacts often result. And when the fundamental disconnect between individual gains and external impacts is coupled with the farmers' bias for near-term cost benefit analysis, as discussed, the harmful sequencing of investments in irrigation and drainage that is observed with notable frequency around the world predictably results. In many arid areas lacking irrigation, farmers often are eager to see the development of a regional irrigation project. Their incentive is clear: with irrigation they can increase production, obtain higher yields, plant a wider variety of crops, and enhance farm income. Their focus generally is on near-term returns, rather than long-term issues regarding salinity or the need for drainage that will arise inevitably at some time in the future. In one sense the farmers' perspective is economically rational. Most people prefer to receive income today, while delaying expenses to the future. Given a positive rate of time preference, most farmers prefer to receive the higher income made possible by developing irrigation, as soon as possible, while delaying the cost of drainage into the future. Clearly, drainage must be provided at some time, but the farmers' initial inclination is to delay the investment, if possible.

15.1.4 Lack of Policy Leadership and Funding by Public Officials Results in Regional Problems

When public agencies require that a viable plan for managing salts and preventing regional waterlogging be implemented at the same time that regional irrigation projects are constructed, they engage in proactive leadership. But public budgets are limited; politicians and agency directors have positive rates of time preference; and waterlogging and salinity problems are delayed outcomes that will not arise for some time. As a result, public investments in adequate drainage systems are often delayed, and policies that might encourage more efficient irrigation or require farmers to manage salts are not implemented in time to prevent regional problems from developing. When the problems eventually require action by the agencies responsible for managing land and water resources or maintaining environmental quality, the technical, financial, and political challenges of implementing regional solutions can be extraordinarily substantial (Wichelns and Oster 2006).

One notable historical-example of proactive public policy leadership in salinity and water management is the San Luis Authorization Act, signed by President Eisenhower in 1957, which authorized construction of an irrigation and drainage system to service lands in the Westlands Water District in California. Irrigation service commenced before the drainage system was constructed, but farmers paid an additional $0.50 for each acre-foot of water they purchased from the U. S. Bureau of Reclamation, to create an account for providing drainage service. The construction of the San Luis Drain and several collector lines was the initial component of a larger drainage system that might have been constructed, over time, if not for the environmental issues that arose in the 1980s

However, the San Luis Authorization Act is not the norm. Government or political failure to prevent salinity buildup arises most often because public officials typically make decisions that involve individual gains and external impacts, rather than grappling with predictable, but delayed, long-term regional impacts. Moreover, public officials have conflicting, non-trivial rates of personal (and political) time preference. Elected public officials must consider the near-term concerns of residents within their voting districts, thus limiting both the time and geographical dimensions of their decisions. For example, many residents of California living near the San Francisco Bay raised substantial environmental issues regarding the potential impacts on water quality if a valley-wide drain were constructed, enabling the discharge of agricultural drainage water directly into the Bay. Their concerns weighed heavily in the state's decision to withdraw support for the proposed, valley-wide master drain (Robie 1988).

In summary, regional salinity and waterlogging problems arise because individuals and organizations lack sufficient incentives to "take the long view" described by Professor Solow. Farmers and public agency personnel tend to focus on near-term returns and rewards and on inputs and outputs to which property rights are clearly defined. In addition, the difficulty of assigning rights to shallow aquifers and measuring salt discharges from irrigated fields discourages agency personnel from implementing policies that might require farmers to minimize the off-farm impacts of their irrigation decisions.

15.2 Three International Case Studies

There are numerous examples, worldwide, of regional waterlogging and salinity problems that have arisen as a result of the three complex factors identified previously: technical relationships, institutional issues, and government intervention.

Some of the cases involve small irrigation and drainage schemes, while others involve very large schemes serving millions of small-scale farmers. The three examples featured in this chapter illustrate the variety of ways in which waterlogging and salinity problems have developed and highlight how farmers, national governments, and international agencies, such as the World Bank, have responded in three distinct settings: (1) the Indus River Basin in Pakistan, (2) the Murray-Darling River Basin in Australia, and (3) the Southeast Anatolia Project in Turkey.

> Much of the world's irrigated land suffers from drainage problems, and an estimated 20 to 30 million hectares need improved drainage. The resulting waterlogging and salinity due to rise of water tables and accumulation of salts are reducing water productivity over wide areas and leading to significant social and economic losses for individuals, households, local communities, and countries. (Ward et al. 2006, p. 166).

15.2.1 Intractable Salinity and Waterlogging in Pakistan

15.2.1.1 World's Largest Irrigation System

Most of the cultivated land in Pakistan is irrigated, and much of the irrigated land is located in the Indus Basin Irrigation System (IBIS), considered by many observers to be the largest contiguous irrigation system in the world. Within the IBIS, farmers divert an estimated 172,000 million m^3 of water from the IBIS Rivers annually to irrigate about 16 million ha (6.5 million acres) of land (Aslam and Prathapar 2006), of which about 9 million ha (3.6 million acres) can be irrigated year-round (Khan et al. 2006). The Indus River and its tributaries – Kabul, Jhelum, Chenab, Ravi, and Sutlej Rivers – originate in the Karakoram, Hindukush and the Himalayan regions along the north and northeastern borders of Pakistan. The IBIS of Pakistan is home to more than 140 million people. The waters of the IBIS Rivers are diverted through reservoirs and barrages into the main canals, which distribute water through a network of branch canals. The current irrigation network comprises 3 reservoirs, 15 barrages and 45 main canals (Aslam and Prathapar 2006).

15.2.1.2 No Final Discharge Solution

The British constructed the extensive canal irrigation system in the IBIS in the nineteenth century, largely to provide a partial supply of irrigation water to hundreds of thousands of smallholder farmers. The primary goals at the time of development were to expand irrigated area, prevent crop failures, and guard against famine (Jurriens and Mollinga 1996). The decision to develop irrigation, while not constructing a drainage system, seemed reasonable at the time, given the limited financial resources available and the pre-irrigation groundwater levels (Fig. 15.1). However, as the groundwater became saline and reached damaging levels, implementing a drainage solution took longer than expected. Several approaches

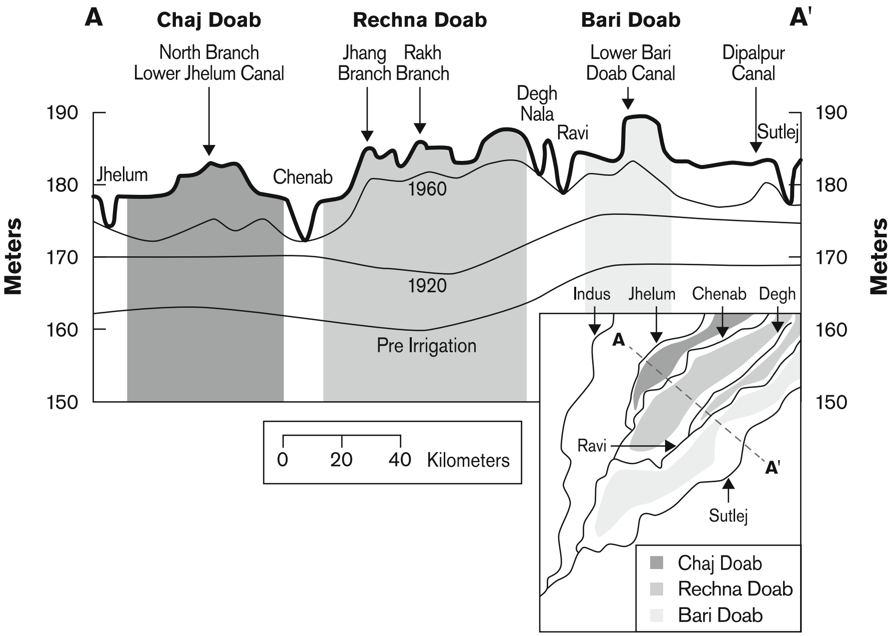

Fig. 15.1 Groundwater levels before (pre-irrigation days of 1920) and after introduction of canal irrigation (1960) in the Punjab Province of Pakistan (Redrawn with permission from Wolters and Bhutta (1997))

were proposed, including vertical and horizontal drainage systems, but, to date, a final discharge site in the Arabian Sea for the region's increasingly saline drainage water has not been approved more than a century after the initial irrigation system was constructed.

At present, the irrigation system in the IBIS operates largely by gravity; many of the canals are earthen; seepage rates are substantial; and farm-level irrigation deliveries are not matched closely with crop water requirements (Qureshi et al. 2008). Since 1960, many farmers in the Indus Basin have installed electric or diesel tubewells to extract groundwater from shallow aquifers to supplement their canal water supplies. Today, farmers using 700,000 private tubewells extract an estimated 24,500 million m^3 per year (20 million acre-foot per year), generating substantial economic benefits for tubewell owners and for farmers who purchase water from the owners (Shah et al. 2003; Khan et al. 2006; Bhutta and Smedema 2007). The tubewells have enabled many farmers to increase productivity in the near term, by improving the reliability and timing of farm-level water deliveries (van Steenbergen and Oliemans 2002). Over time, however, inefficient irrigation and the extensive use of saline groundwater have created large areas of saline, sodic, and waterlogged soils (Qureshi et al. 2008).

The annual salt load in the 172,000 million m^3 of good-quality river water (TDS up to 200 mg L^{-1}) from the IBIS Rivers also is substantial, adding an estimated 2.1 tons of salts per ha on 16 million ha (648 million acres) of irrigated land. Unless the

Table 15.1 Cultivated and salt-affected areas of Pakistan, 2003 (Qureshi et al. 2008)

Province	Cultivated area (10^6 ha)	Salt-affected area (10^6 ha)	Proportion salt-affected area (%)
Punjab	12.27	1.23	10.0
Sindh	5.65	3.04	53.8
Northwest Frontier	2.11	0.11	5.2
Ballochistan	1.84	0.12	6.5
Sum for Pakistan	21.87	4.50	20.6

salt added by irrigation with IBIS river water and saline groundwater is removed from the root zone, salinization is inevitable. Today, salinity impacts an estimated 4.5 million ha (11 million acres), or more than 20 % of the 21.9 million ha (54 million acres) cultivated in Pakistan (Table 15.1). The proportion of area impacted ranges from 5.2 % in the Northwest Frontier Province to 53.8 % in Sindh. Estimates of cropland losses due to salinization range from 28,000 to 40,000 ha per year (69,190 to 98,842 acres per year). The estimated annual financial loss is $230 million (Aslam and Prathapar 2006).

> These serious environmental problems have become a great challenge to ensure food security for the ever increasing population of Pakistan. (Qureshi et al. 2008)

Salinity and waterlogging are particularly damaging from a socioeconomic perspective in Pakistan, where 75 % of the population earns a living through some connection with agriculture, which accounts for an estimated 50 % of gross national product (Qureshi et al. 2008). In addition to reducing crop yields and food security, constraining household incomes, and depressing rural economic development, the hardships caused by salinity buildup and waterlogging of arable land motivate many rural residents to migrate to urban areas. Ideally, increases in agricultural productivity would enable farmers to increase their incomes over time, invest in farm and non-farm enterprises, and gradually create a thriving rural economy.

15.2.1.3 Government Intervention

The government of Pakistan, with the assistance of international agencies, has implemented programs to reduce the extent and severity of salinity and waterlogging, but a fully successful solution has not yet been developed. Within the framework of the Salinity Control and Reclamation Projects (SCARPs), the Water and Power Development Authority (WAPDA) of Pakistan constructed many deep tubewells in the 1960s and 1970s to enhance groundwater supplies for irrigation and reduce pressure from rising water tables. The total cost for the 57 SCARPs is estimated at $435 million (Pakistan Rupee 26.48 billion). More than 20,000 tubewells have been installed on 7.81 million ha. In addition, subsurface tile drains have been installed on 0.22 million ha (0.54 million acres) (Aslam and Prathapar 2006). The SCARPs were successful, for a time, but the cost of operation and maintenance became burdensome, and excessive pumping in some areas allowed saline water to intrude into freshwater aquifers. Currently, the government is reappraising the SCARP program in the IBIS with regard to its (1) finance and administration, (2) environmental impacts, (3) deficiencies in policies and institutional aspects, and (4) inconsistent performance. Important challenges preventing full success of the SCARPs include the high costs of installation and maintenance, the inability of most farmers to share those costs, and the lack of a safe, low-cost discharge site for the saline drainage water (Qureshi et al. 2008).

In addition to the SCARPs, additional technical and institutional approaches are underway or planned to reduce the extent and severity of waterlogging and salinity in Pakistan. With assistance from international donors, the Left Bank Outfall Drain project was completed in 1995 at a cost of $636 million, or $1,230 per ha of land served (Qureshi et al. 2008). The project, which includes nearly 2,000 km of surface drains, 2,000 tubewells, 350 scavenger wells, many other structures, and improvements in irrigation water supplies, has notably reduced water tables and enabled farmers to increase cropping intensities and achieve higher yields (Ali et al. 2004). A similar project is planned for providing drainage service on the right side of the Indus River, but concerns regarding finance, administration, and environmental protection have delayed implementation.

15.2.1.4 National Drainage Program

The National Drainage Program (NDP) was launched in 1997 with the following two objectives: (1) minimize drainage water volume, and (2) facilitate eventual disposal of the saline drainage water from the IBIS to the Arabian Sea. With funding from the World Bank, the work undertaken by the NDP is expected to continue for 25 years, until 2022. Major activities will include the following (Bhutta and Chaudhry 1999; Aslam and Prathapar 2006):

- Repairing and extending existing surface drains (10,000 km),
- Rehabilitating and replacing 1,150 saline groundwater wells,
- Installing pipe drains on 100,000 ha in new areas,
- Lining 1,050 watercourses in saline groundwater areas,
- Constructing interceptor drains (400 km),
- Installing 310 mobile pump stations,
- Reclaiming 16,000 ha of waterlogged-areas through bio-drainage,
- Transferring 1,500 tubewells installed in fresh groundwater areas to farmers,
- Rehabilitating and modernizing canal commands in pilot areas (one/province).

15.2.1.5 Case Study Assessment

Total returns on the initial investments and annual operating costs of the large-scale drainage service projects in Pakistan can be enhanced by improving the institutional framework in which irrigation and drainage are conducted, and motivating farmers to consider the regional, off-site impacts of their irrigation decisions. Water user associations and provincial irrigation and drainage authorities might be helpful in coordinating farm-level efforts to improve irrigation practices and maintain irrigation and drainage facilities (Qureshi et al. 2008). Farmers might be required to pay a portion of the operation and maintenance costs of regional drainage systems, perhaps according to a payment program that provides financial incentives for improving irrigation efficiency and minimizing surface runoff and deep percolation.

Given the large size of the IBIS and the small-scale nature of many rural households, implementing effective solutions to overcome the persistent problems of waterlogging and salinity will be challenging. In addition to the large number of farmers, there is substantial variability in farm practices and hydro-geological conditions across the region. Yet, given the steadily increasing population of Pakistan and the need to stimulate notable economic growth, while maintaining food security, effective long-term solutions must be designed and implemented in the very near future to ensure the sustainability of irrigated agriculture in Pakistan.

15.2.2 Salinity Targets and Tradable Credits in Australia

> With the exception of global climate change, the sustainable management of the Murray-Darling Basin is the biggest single environmental and resource policy issue facing Australia at present. (Adamson et al. 2007).

Agricultural activities in the Murray-Darling River Basin generate more than half the gross value of Australia's crop production and one-third the gross value of its livestock production (Goesch et al. 2007). About three-fourths of Australia's irrigated cotton area and all of its irrigated rice are found in the Basin. Water volume and quality have been major policy issues in the Basin for many years. Surface water supplies are insufficient to meet all demands; much of the groundwater is saline; and the salt loads in irrigation return flows degrade water quality in lower reaches of the river system (Quiggin 1988; Heaney and Beare 2003).

15.2.2.1 History

The historical development of salinity and waterlogging problems in the Murray-Darling River Basin is somewhat similar to that in other regions. Public officials observed the formation of saline high water tables, caused by excessive irrigation, as early as 1912 (Harris 2007). Observations and reports accumulated during the 1920s and 1930s regarding excessive irrigation, salinity, and waterlogging, yet public agencies took little action to encourage efficient irrigation or provide effective drainage service. Construction of drainage diversion schemes to reduce in-stream salinity did not begin until the 1960s.

By 2007, farmers had installed about 90,000 ha (222,395 acres) of subsurface drainage systems in the Murray-Darling River Basin, primarily serving irrigated perennial horticultural crops and pasture (Hornbuckle et al. 2007). Many of the systems discharge substantial salt loads in streams and rivers. In one irrigation district, subsurface drains serving only 7 % of the area discharge 30 % of the salt load leaving the area. In addition to removing salts from the crop root zone, many drainage systems collect and discharge salt that has been stored for millennia beneath the root zone (Christen et al. 2001). Discharging geologic salt increases the salinity of receiving waters, without improving crop production.

15.2.2.2 Institutional Action

The Murray-Darling Basin Commission established a program of tradable salinity credits in 1989. The goal was to encourage the states of New South Wales and Victoria to implement drainage water diversion schemes that would help reduce in-stream salinity at the town of Morgan in South Australia by 10 %, or by 80×10^{-3} dS m^{-1} (Blackmore 1995; Sturgess 1997). By 1994, the average electrical conductivity at Morgan had been reduced by 67×10^{-3} dS m^{-1}, due partly to the states' diversion efforts and partly to changes in river management.

As in many arid areas, the increasing withdrawal of river water for agricultural and municipal uses placed upward pressure on in-stream salinity levels in the Murray-Darling River Basin during the 1980s and 1990s. Diversions increased by 7.9 % from 1988 to 1994, and an audit of the system projected a potential further increase of 14.5 % in the future, in the absence of any restrictions on new water withdrawals (MDBC 2012). This outlook motivated the Murray-Darling Basin Ministerial Council to implement a permanent cap on diversions of water from the Basin, effective July 1, 1997. The cap is viewed by some as providing a necessary, but not sufficient, condition for achieving sustainable management of environmental amenities in the Basin.

The cap on water diversions has generated increased interest in water trades, particularly those involving seasonal water deliveries within river valleys, as farmers and other water users wishing to expand their activities must obtain water supplies from willing sellers (Beare and Heaney 2001). Permanent sales of water allocations across valleys are allowed in concept, but such trading is limited by public concerns regarding regional economic impacts and the possibility of leaving some irrigation districts with inadequate revenues to pay their fixed costs. Inter-valley trades might also increase in-stream salinity levels, if irrigation water is moved from areas with moderately saline return flows to areas with highly saline return flows. Conversely, salinity levels might be reduced if irrigation water is moved from highly saline to moderately saline areas. Perhaps the morals of the water trading story are that (a) a public agency response was necessary to provide a long-term perspective and (b) public agencies might need to monitor and manage water market transactions carefully to ensure that in-stream salinity is not degraded as a result of voluntary water trades within or across river basins.

An updated Basin Salinity Management Strategy was adopted for the Murray-Darling River Basin in 2001. That plan combines engineering solutions, such as drainage diversion schemes, with land and water management planning, and salt disposal within regional subareas, in an effort to further reduce and maintain lower salinity levels at Morgan (MDBC 2006). The Strategy maintains the tradable salinity credits program, which prevents any construction or other activity that would generate a net increase in salinity (Young and McColl 2008).

15.2.2.3 Case Study Assessment

Many of the programs implemented to address water scarcity and salinity in the Murray-Darling River Basin were designed for a climate with higher rainfall than the region has received in recent years. There is notable concern in Australia that a new management regime is needed in the Basin, given the recent, long-term drought and the expectation of a much drier climate in the future (Young and McColl 2008). Current water trading rules might be refined to provide more accurate accounting of available water during extended dry periods, while the opportunity for individuals and consortia to earn and trade salinity credits likely would be retained. Policy discussions involving the Murray-Darling River Basin are quite active at this time, given the substantial reductions in irrigation water supplies made necessary by the recent drought (Smith, 2009, Deputy Chief of Land and Water Division, CSIRO, Canberra, Australia, Personal communication). New policies and guidelines regarding water allocations, water transfers, and environmental quality in the Basin might have notable implications for salinity management.

15.2.3 Investing in Irrigation in Southeastern Turkey

> The multi-sectoral Southeastern Anatolia Project (GAP) is the largest regional development plan for one of the less developed parts of Turkey, with a total area of 7.4 million ha and irrigated area of 1.7 million ha, embracing the sectors of agriculture, industry, energy, transportation, telecommunications, health care, and education. (Aküzüm et al. 1997).

Turkey is home to one of the newest regions to experience salinity and waterlogging problems as a result of irrigation development. The primary goals of the Southeastern Anatolia Project (GAP) in Turkey are typical of such projects worldwide – to enhance economic development and improve livelihoods in a region where 70 % of the economically active population is engaged in agriculture (Ünver 1997a, b). The region includes nine provinces and accounts for about 10 % of Turkey's surface area and about the same portion of its population. In 1985, the region accounted for 4 % of Turkey's Gross National Product, and per capita income in the region was just 47 % of the national average (Ünver 1997b). The 1990 census for the region describes a population of 5.15 million, with an annual growth rate of 3.4 % (Altinbilek 1997). The high population growth rate, extensive poverty, and lack of employment opportunities have contributed to the net out-migration of residents in all nine provinces.

Economic conditions have been declining for many years in Southeastern Anatolia, while increasing in other regions of Turkey. One goal of the GAP project is to reverse the trend of increasing differences between income levels in Southeastern Anatolia and those in western portions of the country (Aksit and Akcay 1997). The notion of boosting economic activity in Southeastern Anatolia through large-scale investments in irrigation was first considered in the 1930s (Ünver 1997a). Since then, and prior to the onset of irrigation deliveries in 1995, the project has become a major economic development program involving complementary investments in several sectors (Ünver 1997c).

15.2.3.1 GAP Irrigation Project

The soils in Southeastern Anatolia are fertile, but annual rainfall is limited, ranging from 1,200 mm in the north, to just 300 mm near the Syrian border in the south (Aküzüm et al. 1997). The primary constraints to enhancing agricultural production are inadequate rainfall and the sub-optimal geographic distribution of rainfall during the growing season (Altinbilek 1997). Given the large portion of the population engaged in agriculture and the potential improvements in productivity that often accompany investments in irrigation, many academics and public officials supported the GAP project concept (Saysel et al. 2002). The officially stated goals of the project were broad and comprehensive:

- Increase agricultural production,
- Improve income levels and stimulate capital accumulation in agriculture,
- Enhance the ability of urban areas to accommodate the region's increasing population, and
- Promote social stability and sustainable economic development.

To achieve these goals, the GAP project includes major investments in irrigation, some of which are similar to those provided by the U.S. Bureau of Reclamation and State of California in the San Joaquin Valley. With investments in hydropower, agriculture, urban infrastructure, forestry, health care, and education, the GAP has become the largest regional development project undertaken in Turkey (Kulga and Çakmak 1997; Ünver 1997c).

In the beginning, the GAP project was viewed with notable enthusiasm and great potential. Writing shortly after irrigation deliveries began in 1995, Akuzum et al. (1997) suggested the "GAP will double Turkey's hydroelectric production, increase irrigated areas by 50 %, more than double per capita income in the region and create two million new jobs in the coming decade." Ünver (1997c) predicted that cropping intensity would increase from 89 to 134 %, cotton area would increase from 2.8 to 25 %, and the area planted in wheat, barley, and pulses would decline from 72 to 45 %.

15.2.3.2 Salinity and Drainage Issues

With the GAP Project underway, potential salinity problems were not the focus of attention, perhaps because salinity and drainage had not been problems during centuries of rain-fed production in the region. Quoting from Ünver (1997c), "Salinity and alkalinity problems are minimal, and most of the soil has good drainage conditions." Enthusiasm and optimism are desirable characteristics pertaining to major investment projects; however, the long historical record of salinity and drainage problems arising in arid regions after investments in large-scale irrigation schemes might have generated greater concern, analysis of their similarity to potential future conditions in Turkey, and proactive planning among officials designing and implementing the GAP project (Tekinel et al. 2002).

Within 12 years of the onset of irrigation deliveries, salinity and drainage problems began limiting agricultural productivity in Southeastern Anatolia. By 2004, an estimated 225,000 ha (555,987 acres) were being irrigated with GAP project water deliveries (Adaman and Özertan 2007). The project is expected to provide irrigation for 1.7 million ha (4.2 million acres) upon completion. To date, the GAP project has enhanced economic output from the region, generated new employment opportunities for residents (Kendirli et al. 2005), and attracted migrant workers from other regions during harvest seasons (Kudat 1999). Much of the increase in economic activity pertains to cotton production. While the original plan for the GAP project suggested that cotton eventually would be planted on 25 % of the irrigated area, farmers already have expanded cotton production to cover 90 % of the irrigated area. The climate, infrastructure, and market conditions favor cotton production from the farm-level perspective.

Many farmers began growing cotton for the first time when irrigation water became available via the GAP project. Lacking adequate knowledge of cotton production and water management, many farmers over-irrigated their cotton fields, causing excessive deep percolation, rising water tables, and soil salinization. For examples, farmers applied estimated 4,242 and 3,241 mm of water that required estimated 611 (in 1998) and 592 (in 1999) mm, respectively, more than five times the water required for successful crop production. By the end of 2004, salinity problems were observed on about 15,000 ha (37,066 acres) in Harran, and an estimated 40,000–50,000 ha (98,842–123,553 acres) were threatened by rising water tables. In addition to increasing soil salinity, the potential harm from rising water tables in the region includes damage to fruit trees, roads, buildings, and the drinking water system (Adaman and Özertan 2007).

In a survey of 619 farm households on the Harran Plain in 2005, Adaman and Özertan (2007) observed that 75 % of the farmers knew of the relationship between irrigation and salinity, and 91 % of farmers were aware of salinity problems on their own fields. Nonetheless, about one-half of the farmers interviewed were not aware of the full extent of salinity problems on their fields, and only 12 % had received any training with respect to salt management within the previous 2 years. Almost three-fourths of the farmers were willing to engage in collective action to remedy the salinity situation in the region, but it was not clear how such action might be organized. The authors conclude that the following factors have contributed to the salinity and drainage problems on the Harran Plain:

- Irrigation deliveries were started without first building a proper drainage system.
- Furrow irrigation, a rather inefficient method, is used extensively.
- Excessive amounts of irrigation water have been applied, due partly to the very low incremental cost of water and the lack of training with respect to efficient irrigation methods.
- Cotton has become the dominant crop.
- Farmers have planted too few halophytes that might be helpful in preventing saline water from accumulating near the ground surface.

15.2.3.3 Lack of Farmer Training in Irrigation Methods

While the estimated cost of infrastructure required to irrigate farmland in the region is \$4,600/ha (\$1,862/acre), less than 1 % of that amount is invested in educating and training farmers (Yildrum 2004, cited in Adaman and Özertan 2007). Adaman and Özertan (2007) find the inadequate training to be unfortunate, given that farmers can avoid excessive deep percolation by using efficient irrigation methods. Farmers on the Harran Plain who received extensive training regarding salinity reduced the frequency of irrigation events from 10 to 15 times per season to 6–7 times (Kün et al. 2005, cited in Adaman and Özertan 2007).

15.2.3.4 Case Study Assessment

The current situation regarding irrigation, drainage, and salinity on the Harran Plain is very similar to the situation that has prevailed in many other arid regions throughout history. The Government of Turkey is eager to promote economic development, provide jobs, and enhance food security in the Southeastern Anatolia region. Irrigation can stimulate economic development by enabling farmers to produce a larger assortment of crops, while obtaining higher yields than are possible without irrigation. Given positive rates of time preference on the part of farmers, irrigation planners, and public officials, the irrigation components of the GAP move forward on an aggressive schedule, while installation of the drainage components are delayed. Eventually, investments and policies that encourage wiser irrigation and drainage management will be needed to sustain the benefits provided by irrigation on the Harran Plain.

The causes of salinity and drainage problems on the Harran Plain in Turkey are similar to those pertaining to the San Joaquin Valley in California. There, too, irrigation systems were built before drainage was provided, and farmers used surface irrigation methods to apply low-priced water to large fields of cotton and other crops, often while generating large amounts of deep percolation. As groundwater levels rose, and salinity problems became apparent, drainage eventually became recognized as an urgent issue requiring substantial regional investments. In California, unlike Turkey, the selenium issue (a toxic trace element contaminant) at Kesterson Reservoir required public officials to act quickly to prevent harm from selenium in agricultural drainage water. In both locations, long-term, sustainable solutions to salinity and drainage problems are required to prevent continuous decline in agricultural productivity.

15.3 Lessons California Learned

Policies that support irrigated agriculture on a national or regional scale are difficult to sustain after problems resulting from the need for drainage become well entrenched, or when the availability of water becomes limiting. In the United States, the increased concern about environmental impacts of irrigation and the toxicity due to selenium at the Kesterson Reservoir occurred at about the same time. How the issues of irrigation sustainability and environmental protection may play out in California will be a unique contribution to the international body of knowledge about the sustainability of irrigated agriculture, because it was the first situation where potential environmental catastrophe caused by irrigation involved the toxic trace element, selenium. Also unique are the efforts by the state and federal governments to build a regulatory system that facilitates both irrigation sustainability and environmental protection (Johnston et al. 2011). The following sections address these topics.

15.3.1 Era of Regulatory Control

15.3.1.1 Tulare Lake Basin – Southern San Joaquin Valley: Disposal of Drainage Water into Evaporation Basins

The Tulare Lake area is a closed basin with no natural outlet. The State of California originally planned to develop a Master Drain jointly with the federal Central Valley Project service area. The drain would have extended from the southern portion of the Tulare Lake Basin to the Delta. When the State of California abandoned this effort, farmers and water districts in the southern end of the Tulare Lake Basin realized that they would need to manage and dispose of their own drainage water within the Basin in evaporation ponds. The Central Valley Water Quality Control Board (CVWQCB) established requirements under the State Porter-Cologne Water Quality Control Act to regulate the construction and operation of evaporation basins, which included the following conditions:

- Total containment of the drainage water in the basins,
- Flood protection provided for a 100-year flood event,

- Use of a multi-celled salt routing design,
- Provisions for access by Department of Fish and Game officials to monitor wildlife,
- Provisions for groundwater protection by interceptor drains or other means.

Under this policy, by the early 1980s, there were 27 separate evaporation basins constructed and operating in the Tulare Lake Basin. Of these, as of 2010, only six evaporation ponds in Tulare Lake Drainage District are in operation. This reduction was a consequence of the selenium problems at Kesterson Reservoir. The CVWQCB found selenium in all the inflows to evaporation basins in the Tulare Lake Basin. Consequently, new management requirements were put in place.

The new requirements focus on discouraging wildlife use of the ponds and/or providing mitigation and compensation habitat for any unavoidable losses. To comply, the pond operators took various steps to modify pond design and operation (Chap. 9). Some of the most successful were the following:

- Steeping the inside-slopes of the individual basins to at least 3:1 to discourage shorebird feeding,
- Keeping the entire pond area and banks free of vegetation,
- Maintaining a minimum water depth of at least 60 cm (24 in.)
- Removing windbreak islands that attracted nesting birds due to their isolated location,
- Removing man-made bank stabilization materials, such as automobile tires and concrete chunks, that were used for nesting and shelter,
- Removing immediately any sick or dying birds to keep avian diseases under control,
- Hazing of migratory waterfowl attracted to the ponds,
- Monitoring compliance of drain water inflows and pond water and sediment for selenium and salinity, and
- Monitoring of birds for abundance and any signs of toxicity.

Because of the unavoidable bird losses in some of the basins due to selenium, the CVWQCB also required that compensation wetland habitat become a part of the management of these basins. The mitigation measures implemented on the evaporation basins proved to be effective in reducing the nesting and foraging of water birds. The performance of the constructed compensation habitat shows that proper design and operation of such habitat can promote shorebird safety and abundant reproduction (Davis et al. 2008).

15.3.1.2 Grasslands Watershed – Disposal of Drainage Water into the San Joaquin River

The experience of the Grassland Water District in the San Joaquin Valley to cope with the salinity and selenium (Se) problems is discussed in detail in Chaps. 1 and 2. In order to protect wildlife and aquatic life in wetlands, in areas such as occur in the

Grasslands watershed, the federal and state governments established a water quality objective for Se of 5 μg L^{-1} (ppb) (CVRWQCB 1988, 1996). The State of California set a lower limit of 2 μg L^{-1} (ppb) in surface waters because Se concentrations increase as water evaporates, particularly where water flow is slow, as in wetlands. In 1988, the CVRWQCB set out a program to begin to control the agricultural subsurface drainage discharges from the Grassland watershed (CVRWQCB 1988). Several policy actions were taken, including the following:

- The control of Se in the drainage water was set as the first priority;
- The San Joaquin River could continue to be used to remove salts from the basin, provided water quality objectives for Se were met;
- Any further increase in drainage water discharges to the San Joaquin River from the Grassland watershed were prohibited until load levels were shown to be within the water quality objectives;
- Regulation of Se discharges would be pursued on a regional basis, rather than on individual farms;
- Reuse of drainage water would be encouraged; and
- A separate and isolated valley-wide facility to take drainage water out of the basin would continue to be promoted as the best long-term alternative.

These policy actions were supplemented with a program of implementation that relied on voluntary actions by individual farmers to improve their irrigation efficiency with the goal of reducing the Se load that each farm was discharging. The program had mixed success, primarily because individual farmers did not know the consequences of their actions – how increasing irrigation efficiency would improve downstream water quality – and there was no overall responsible party who could direct the voluntary actions of individual farmers or verify results (Westcot et al. 1996; Engberg et al. 1998). Also, when there are multiple sources of Se in irrigation drainage water, the actions of any one farmer may not have had a significant impact on water quality.

Regulatory action begun in 1988 to reduce Se loads entering the San Joaquin River focused on farm water conservation to reduce deep percolation and thus reduce Se mobilization into the drainage water. However, an extended drought that began at about the same time, provided a strong incentive for water conservation and improved irrigation efficiencies. The reduction in drainage water discharged into the San Joaquin River resulted in a 50 % reduction of Se in the river water (Karkoski 1994; CVRWQCB 1996). However, the Se concentrations in the water in the internal channels within the wetlands of the Grassland watershed increased. With the return of the wetter cycles after 1993 and a full water supply to each farm, the total load of discharged Se returned to the 1988 levels.

In 1996, the regulatory agencies (CVWQCB, SWRCB) established a new approach to manage Se based on source control, including lining irrigation canals, improved on-farm efficiency; on-farm reuse of drainage water, no discharge of subsurface drainage water into channels that supply the wetland areas, and control of discharges into the San Joaquin River. The related regulations were promulgated under a formal Waste Discharge Requirement (state equivalent of a permit) with specified monthly and total annual load limits to specific water bodies. This was the first time a permit-type approach was used to control or regulate a non-point source discharge. The permit was issued to a responsible regional entity, the Drainage Authority of the San Luis Delta Water Authority, which has the administrative power to implement the load limitations. A regional entity is required to develop long-term solutions that require complicated control mechanisms (Young and Congdon 1994).The new regulatory policy recognizes the difficulty in meeting water quality objectives and that not all actions can be accomplished at the same time. Wetlands and wetland water-supply channels were given the highest priority for protection, followed by protection of in-stream aquatic life in the San Joaquin River and finally by protection of in-stream aquatic life in the 10 km long channels where the inflows are dominated by drainage water (CVRWQCB 1996; Westcot et al. 1996).

Similar regulatory control policies were put in place to protect aquatic life in the San Joaquin River (Karkoski et al. 1993; Karkoski 1994; CVRWQCB 1996). Selenium and salt limits were established for the entire watershed, which includes the Grassland Watershed. This approach allowed greater participation by the on-farm drainers and their water and drainage agencies in deciding how to apportion loads among themselves to achieve the most cost-effective methods for compliance (Young and Congdon 1994).

To meet the Se load limitations, farmers and water district managers have implemented the most aggressive source control and drainage management program ever conceived (Quinn et al. 2006).

15.3.2 Investments and Institutional Reforms Are Needed Worldwide

Substantial investments are needed worldwide to improve irrigation and drainage management on existing agricultural lands, to provide the food and fiber needed to support a world population that likely will increase by 50 % by 2050 (Schultz et al. 2005; Molden 2007). National governments and international donors must ensure that sufficient funds are available for conducting research, designing new interventions, and implementing innovative irrigation and drainage programs around the world. Existing drainage systems need replacement on an estimated 30 million hectares (74 million acres), while new systems are required on another 30 million ha (74 million acres) (Ritzema et al. 2007). About one-third of these 60 million ha (148 million acres) are in Egypt, India, and Pakistan, countries in which most residents are poor, and most earn their living through some connection with agriculture.

The estimated costs of new drainage systems are quite high and are likely beyond the affordability of many small-scale farmers. Drainage systems can be installed for about $800–$1,600/ha ($324–648/acre) in the eastern United States, if a gravel envelope is not needed and there are no costs of obtaining a drainage outlet (Skaggs et al. 2006), The estimated cost of $5,000/ha ($2,023/acre) in California includes the expense of a gravel envelope and a field-level collector system capable of delivering drainage water to a regional collector system (Wichelns and Oster 2006). Installation costs in developing countries likely will be on the low end of this wide range, given the lower costs of labor and other materials. Even so, the cost will be prohibitive for most small-scale farmers.

Agricultural incomes are limited in many of the areas requiring drainage investments, due partly to the current, degraded state of agricultural lands, and partly to farmers' limited access to affordable, complementary inputs, such as high-quality seeds, fertilizer, and farm chemicals. The optimal approach to improving drainage conditions will include comprehensive financial analysis of near-term investments and expenditures in drainage systems, near-term investments in other forms of infrastructure, expenditures for complementary variable inputs, and the long-term returns to enhanced agricultural production. Over time, the need for public support should be reduced, as farm-level incomes increase, due to improvements in crop yields, cropping patterns, and market opportunities.

Many researchers are developing new technological approaches to reducing the areal extent and severity of waterlogging and salinity in arid areas. Many others are investigating institutional innovations that might provide the correct incentives for managing irrigation water with greater care at the farm level and for minimizing surface runoff and deep percolation throughout irrigated areas. Combinations of technological and institutional advances likely will be most helpful in restoring agricultural productivity in degraded areas and preventing new areas from becoming waterlogged and saline.

Many rural households in developing countries might benefit from the installation of small-scale drainage systems that provide relief from saline, high water tables at moderate cost. Kahlown et al. (2007) examined the performance of three small-scale drainage systems during a 10-year, farm-level study in the Indus Basin of Pakistan. The systems reduced soil salinity, improved land productivity, and enabled farmers to begin producing high-valued crops, such as rice, cotton, and sugarcane. Farmers were required to provide a portion of the capital cost of installing the systems. They were happy to pay that portion, given the higher incomes they earned as a result of obtaining drainage service.

Khan et al. (2008) present a method for determining the optimal pattern of net recharge to shallow groundwater at the farm level, in areas with regional salinity and waterlogging problems. Public agencies might use this approach to assign responsibility to farmers for improving irrigation management and reducing net recharge. Farmers can use the model to determine optimal irrigation strategies, subject to net recharge constraints.

Further research is needed also regarding the disposal of saline drainage water in ways that are consistent with maintaining environmental quality. Just as the lack of a suitable outlet for drainage water has prevented farmers on the west side of California's San Joaquin Valley from achieving long-term sustainable irrigation, issues regarding the transport and discharge of saline drainage water are notable in many arid areas. Terminal drainage water disposal constraints, similar to the constraint in California, limit irrigation and drainage development in Armenia, Egypt, and Turkey (Smedema and Shiati 2002). In Pakistan, farmers in the Sindh Province prefer that drainage water collected in the Punjab not be discharged into the Indus River, which flows from the Punjab through Sindh on its path toward the Arabian Sea (Bhutta and Smedema 2007; Qureshi et al. 2008).

Plans for salt disposal within irrigated regions need to be considered carefully (Oster and Wichelns 2003; Rhoades 1989), to ensure that projects reduce impacts downstream, provide incentives to use irrigation water sparingly, and foster development of community efforts that fully acknowledge hydrologic linkages. Public agencies must consider interactions involving surface and subsurface sources of irrigation water, while developing groundwater management plans that optimize the longevity of water quality by localizing the impacts of in-region salt disposal. Evaporation ponds are not a perfect solution, as leaks will occur over time. In addition, drainage systems cannot capture all drainage water when regional water tables are impacted by groundwater pumping (Chang et al. 1983). Sustainable solutions to waterlogging and salinity problems will include farm-level and regional efforts to minimize drainage water volume and inter-state or inter-provincial agreements regarding the suitable transport and disposal of drainage water, or the salt it contains.

15.4 Conclusion

There are both universal causes of salinity and drainage problems and some relatively universal approaches to their solution. The experience in the San Joaquin Valley is remarkably similar to that in other drainage-impaired regions. Government, industry, and individuals initially cooperated to develop land and water resources, often with subsidized infrastructure. There was a temporal lag between the start of irrigation and the first signs of potential salinity and drainage problems. Following the discovery of a problem, several methods of managing irrigation and drainage were adopted, based on the known science and engineering. A regional or national government led the initial effort to improve water management, as such entities have a longer-term perspective on costs and benefits and they have the financing and jurisdiction to implement solutions at the appropriate spatial scale. Ultimately, the solutions have been site-specific and they have usually resulted in new research efforts to better understand the underlying chemical and physical processes.

In this sequence of events, research and development play critical roles; they form the bridge between problem and solution. Recognizing this, public programs to develop or expand irrigated agriculture must provide for early monitoring of irrigation and drainage impacts and must support research to address potential problems. Lacking knowledge, and following common practice, creates the risk of repeating mistakes made in the past. Yet knowing what has happened before does not prevent making the same mistakes again. Continuous research, proper incentives, and the appropriate time horizon are essential in achieving sustainable use of land and water resources.

As is evident throughout this book, technologies already exist to manage both the salinity and Se problems on-farm and at the regional scale in California. Environmental impacts of salts and trace elements can be managed at acceptable levels. Both applied and basic research continues to improve these technologies. Scientists are investigating institutional innovations that might provide the correct incentives for managing irrigation water with greater care at the farm level and for minimizing surface runoff and deep percolation throughout irrigated areas. Combinations of technological and institutional advances will need to continue to assure capabilities exist to handle new problems and issues that will continually emerge in the future.

We have placed a current perspective on Professor Hilgard's recognition in 1886 that sustainable irrigation management requires community involvement and collective efforts. He would add the caveats that irrigation needs to use water sparingly and drainage is a necessity (Hilgard 1886).

Acknowledgement We wish to thank the following for their help in writing this chapter: Professor Mehmet Ali Cullu, University of Harran, Department of Soil Science, Sanluerfa, Turkey; Dr. Manzoor Qadir, ICARDA, Aleppo, Syria; Dr. Chris Smith, CSIRO, Canberra, Australia; Dennis W. Westcot, formally California Regional Water Quality Control Board, Central Valley Region, Sacramento, California; Douglas E. Davis, Tulare Lake Drainage District, Corcoran, California, Michael L. Delamore, formally U.S. Bureau of Reclamation; William (Bill) R. Johnston, formally Westlands Water District, Fresno, California; Sarge Green, Westside Resource Conservation District, Fresno, California; and David Cone, Southern San Joaquin Valley Water Quality Coalition, Fresno, California We also thank Debra Silva for her help with text editing.

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Index

A.C. Chang and D. Brawer Silva (eds.), *Salinity and Drainage in San Joaquin Valley, California: Science, Technology, and Policy*, Global Issues in Water Policy 5,
DOI 10.1007/978-94-007-6851-2, © Springer Science+Business Media Dordrecht 2014