TREATMENT WETLANDS

SECOND EDITION

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Preface

Since the first edition of this book, treatment wetland technology has advanced on all fronts. Considerably more is known today about how treatment wetlands function. Over the last decade, wetland technology has evolved into new reactor configurations, a much broader range of treatment applications, and a dramatically expanded presence worldwide.

This growing knowledge base leads to an increased appreciation of just how complex treatment wetlands are. Because treatment wetlands are strongly influenced by a variety of internal and external ecological cycles, the assumptions that simplify the analysis of conventional reactors in the environmental engineering field can no longer be justified. As wetland technology continues to evolve, much effort is being applied to understand both short-term and long-term variability cycles within treatment wetlands. Because treatment variability strongly influences wetland design, factors that influence performance—especially the role of internal biogeochemical cycles—become paramount in understanding how treatment wetlands function. This knowledge can then be applied to make informed decisions regarding wetland design.

WHAT IS A WETLAND?

The meaning of the word *wetland* has been severely stretched in the treatment wetland literature. We would generally insist that wetlands have plants, water, and some kind of media. Without plants they are soil, sand, or gravel filters, or ponds. In fact, planted gravel filters— meaning all subsurface flow wetlands—have no natural wetland analog. Similarly, it is not unusual to hear discussion of "treatment wetlands" that do not have plants. We have tried to use commonly accepted terminology for systems that are generally regarded as within the scope of the treatment wetland field. We have also distinguished systems based on their hydrology, which may be horizontal flow, pulse-feed vertical, fill-and-drain, or recirculating.

SCALING FACTORS

Treatment wetlands are in-the-ground, outdoor systems. With other visions guiding them, treatment wetland researchers sometimes find wetlands to be potted plants, pots filled with gravel and no plants, sections of pipe, flasks, test tubes, and all manner of tubs, tanks, and troughs, sometimes with recently inserted propagules. Indoor systems do not experience wind, sun, birds, and animals. When the size is too small, the system is subject to severe edge effects. Although comparative results from small lab systems are useful, there is often the unstated assumption that they would represent the treatment performance of a full-scale wetland. We have tried to be reasonably careful by drawing attention to scale with the terms *microcosm*, *mesocosm*, and *pilot project*.

SHORT-TERM STUDIES

We find too many studies are based on infant or juvenile ecosystems, which have not had time to mature into the full suite of components that occur in fully developed wetlands. We also find too many studies focus on short-term events. This, we believe, is like interpreting the meal-time hamburger intake rate of teenage boys and girls as their sustainable caloric intake.

For instance, the development of bed clogging in HSSF wetlands has not been studied in a systematic way in the academic community. Recent knowledge of bed clogging has come from the hydraulic failure of full-scale systems (often at a high price) because clogging phenomena takes longer to develop than the tenure of a typical graduate student. As a result, the long-term viability, and maintenance requirements, of HSSF wetlands is still unknown, despite the fact that thousands of systems have been constructed worldwide.

It is fortunate that there are now numerous full-scale projects to balance the data scales.

WHAT'S NEW?

Of course, there is much more information available now than in 1995 when the previous analyses were completed. The doubling time of the available data is on the order of two or three years, because old systems continue to return new information as more and more systems come on line in more and more application areas. As a consequence, about 90% of the data used in support of this book was not available at the time of the first edition. It has been reassuring to find that most of the data and interpretations of the first edition have stood up well to the test of time, but not surprising to find that some numerical interpretation had to be updated.

Data analysis in the first edition was predicated on the plug flow assumption, despite the known fact that virtually no treatment wetland actually tested out as plug flow. It is now understood that while this may provide acceptable interpolation on existing performance ranges, it can lead to very bad extrapolations that should not be used in design. Further, it has been recognized that weathering of the mixtures that comprise many of the standard wastewater parameters will also invalidate the plug flow assumption. Accordingly, a mixing parameter has been added to the mathematical representation of wetland behavior.

DESIGN TOOLS

With the advent and proliferation of desktop computing, expectations for calculational detail have risen markedly in the last 15 years. It is no longer necessary to be given a single equation, arranged to be solved for the single variable of interest. This second edition is predicated on the extensive use of spreadsheets, and the large array of iterative and optimization tools that go with them. The scientific design team for a constructed wetland must include that capability, or else be constrained to simple scale-up or scale-down for a repetitive design.

In the first edition, central tendency rate coefficients were presented, along with tables detailing the values for individual systems. Several investigators soon found that their results did not match the central tendencies, and incorrectly concluded that something must be wrong. In this edition, we have therefore opted to present the distributions of rate coefficients across numerous wetlands of all types, so that new results may be placed in that spectrum, and designs may be selected with different positions across the intersystem landscape.

The scatter of wetland outlet concentrations around an often-seasonal trend is another type of variability to be accounted. The first edition utilized maximum monthly deviations across the year. Here, the annual pattern is accounted separately, based on system performances, and various percentiles of the exceedance distribution are presented as a necessary part of design.

Among the differences between the new and the old data interpretations, the narrowing of the gaps between surface and subsurface flow system performance and cost are perhaps the most intriguing. Based on new and greatly expanded data analysis, subsurface flow wetlands do not enjoy much of a performance margin on a per unit area basis, and may be less effective than surface flow systems for some contaminants. However, the cost differential is much less than previously thought, when comparable-sized wetlands are evaluated, but still remains about a three to one capital advantage for surface flow. Therefore, nuisance and health hazard avoidance rules the selection of wetland type.

TECHNOLOGY SELECTION

In the early years of constructed wetland technology, and to some extent continuing today, there was a tendency to consider wetlands as stand-alone devices, usually accompanied by pretreatment. It is now understood that series and parallel natural system networks, perhaps involving recirculation, are sometimes better choices. Combinations of vertical flow, horizontal subsurface flow, ponds, and free water surface wetlands are increasingly being used.

THIS BOOK

This book has been updated to reflect the dramatic advances in wetland technology over the last 12 years. The authors of this second edition come from different backgrounds, and work in different aspects of the treatment wetland field. By combining our knowledge and experience, we have endeavored to present a broad range of information regarding the science, hydrology, hydraulics, reactor theory, applied design, implementation, cost, and O&M of treatment wetland systems.

The format of the second edition reflects a dual approach. Part I is organized in a manner that allows the reader to explore the internal mechanisms by which treatment wetlands operate. Existing projects and operating results from real-world treatment wetlands are utilized extensively. Internal mechanisms, their influence on treatment performance, and their effect on system variability are explored in detail in Part I.

Part II is organized to allow the reader to examine how performance data is analyzed and applied to the design process. Like the first edition, this book adopts a performance-based approach to design, in addition to presenting design tools such as loading charts and scaling factors. Continuing with the theme of practical implementation, Part II also summarizes current knowledge that is key to getting wetland projects built, including construction methods, cost information, and operation & maintenance (O&M) requirements.

We have not repeated the natural wetland fundamentals that are contained in the first edition, nor have we reiterated databases or case histories contained therein. All other topics have been nearly totally rewritten, as required by the vastly increased data sources and understanding that have developed in the many years since the first edition.

However, as much as things have changed, some things remain the same. The predictions made in the first edition about rapid evolution of treatment wetlands have certainly proven true. We expect that, if anything, this rate of change will continue to increase after the publication of this second edition, which might have been more properly called *Treatment Wetlands II.*

Robert H. Kadlec
Scott D. Wallace

Acknowledgments

The authors want to acknowledge our families and friends who supported us while writing this book. For Bob Kadlec, the extreme patience of his wife Kelli was a paramount virtue, as she put up with over a year's worth of working weekends and the virtual loss of a spouse. Scott Wallace would like to thank his coworkers at North American Wetland Engineering, who stepped forward and handled all the challenges of managing projects and running an engineering company so he could have the freedom to write this book. It is a pleasure to work with such a group of excellent people.

A tremendous amount of effort was given by Jan Vymazal, who helped immensely in the preparation of Part I of this book. His broad understanding of treatment wetlands is evidenced in his many authored and edited volumes, and we are very grateful for his assistance.

Jaime Nivala was instrumental in the completion of this book. She carefully reviewed every chapter, figure, and table; her abilities as both coordinating editor and environmental engineer were invaluable to us. Jaime did an excellent job of managing the myriad details of producing a book of this scope, and her organizational skills made the writing process much easier. This book could not exist in its current form without her extraordinary efforts.

We also want to thank Sue Knapp, who injected a breath of life into the cover design and all of the engineering drawings and hand sketches that are now the final artwork in this book.

This book expands upon many concepts advanced in the first edition, for which Robert Knight bears a full share of credit. He was a major architect of the foundation for this work.

The authors wish to acknowledge the efforts of the hundreds of engineers and scientists who have had the courage to create, innovate, and ultimately develop treatment wetlands as a viable technology to solve many environmental problems. The friendly and open communication between colleagues at international conferences has made this field a pleasure to work in, and the "lessons learned" have greatly contributed to the rapid evolution of treatment wetlands.

We are very appreciative of those projects that have shared data with us. Without the data assembled from these diverse resources, this book could not exist. The list is long, and these hundreds of project owners are owed heartfelt thanks for their generosity.

Robert H. Kadlec
Scott D. Wallace

Authors

Robert H. Kadlec holds B.S., M.S., and Ph.D. degrees in chemical engineering from the University of Wisconsin and University of Michigan, 1958–1962. That era saw the culmination of the "unit operations" approach to chemical processing, and the transition to the use of principles of transport phenomena to describe transfer and reaction rates in a wide variety of chemical and biochemical processes. Those techniques and analytical tools are also the foundation of today's environmental engineering. Bob began applying engineering analysis to wetland processes in 1970, with the goal of managing wetlands for water quality improvement. The result was the Houghton Lake natural wetland treatment system, which is still operating successfully.

Research on that natural wetland, and on the ensuing 30 years of its operation for engineered treatment, formed the early framework for Dr. Kadlec's development of wetland process characterization. The technology has grown tremendously, and so has Bob's involvement in treatment wetland projects. He has participated in over 250 projects, ranging from simple feasibility studies to comprehensive university research projects. Early university studies focused primarily on wetland hydrology and water chemistry. In the course of many projects, a good deal of knowledge of practical ecology was imparted by his colleagues.

He has worked on treatment wetlands in many states and several other countries, participating in the design of over a hundred treatment wetlands. Major and long-running projects have included Houghton Lake, Michigan; Incline Village, Nevada; Hillsdale, Michigan; Columbia, Missouri; and the Everglades Stormwater Treatment Areas. He is past chairman of the International Water Association (IWA) Specialist Group on the Use of Macrophytes in Water Pollution Control. He has authored or coauthored over 130 publications on treatment wetlands, in addition to dozens of project reports. He was a proposer and developer of the U.S. EPA North American Treatment Wetland Database.

Dr. Kadlec retired from his teaching duties in 1993, and is currently doing business as Wetland Management Services, providing specialty consulting services to a wide range of governmental and private organizations. His contributions to this book are an effort to consolidate over three decades of research and practical experience.

Scott D. Wallace began his career as a wastewater treatment plant operator, and also worked as a field technician and analytical chemist. He earned a B.S. in civil engineering (1986) and an M.S. in environmental engineering (1989), both from the University of Iowa. Scott has worked full-time as a consulting engineer since 1988, and has been employed at CH2M HILL, Shive-Hattery Engineers and Architects, and HDR Engineering. Scott began designing treatment wetlands in 1992, beginning with the Indian Creek Nature Center, one of the first cold-climate subsurface flow wetlands in the United States.

In 1997, he cofounded North American Wetland Engineering (NAWE), a consulting firm focused on the development and application of treatment wetlands. Since then, he has designed over 200 treatment systems, the majority of which involve wetlands. NAWE was acquired by Jacques Whitford in 2007, and Scott currently works as a principal in the Water Resources Sector for Jacques Whitford. He consults on a wide variety of projects in the United States and internationally.

Scott has been active in research and development, and holds 5 patents on wastewater treatment systems, including aerated subsurface flow wetlands, a technology briefly discussed in this book. He is a registered professional engineer in 22 states, and has written numerous technical papers on treatment wetlands. Scott was the principal investigator for *Small-Scale Constructed Wetland Treatment Systems; Feasibility, Design Criteria, and O&M Requirements,* a design manual published by the Water Environment Research Foundation (WERF) in 2006. He is an active member of the IWA Specialist Group on the Use of Macrophytes in Water Pollution Control.

Part I

Technical Underpinnings

1 Introduction to Treatment Wetlands

Since the first edition of this book, treatment wetland technology has advanced on all fronts. Considerably more is known today about how treatment wetlands function. Over the last decade, wetland technology has evolved into new system configurations, a much broader range of treatment applications, and a dramatically expanded presence worldwide.

This growing knowledge base leads to an increased appreciation of just how complex treatment wetlands are. Because treatment wetlands are strongly influenced by a variety of biological processes and by biogeochemical cycles, the assumptions that simplify the analysis of conventional treatment reactors in the environmental engineering field may no longer apply. As wetland technologies continue to evolve, countless effort is being applied to better understand the short-term and long-term treatment and variability cycles in these systems. Because treatment variability strongly influences wetland design, the factors that influence performance, such as hydraulics and internal biogeochemical cycling, become paramount in understanding how treatment wetlands function. This knowledge can then be applied to make informed decisions regarding wetland design.

The format of this second edition reflects this approach. Part I is organized in a manner that allows the reader to explore the internal mechanisms by which treatment wetlands operate. Operating results from existing treatment wetland projects are extensively analyzed. Internal mechanisms, their influence on treatment performance, and their effect on system variability are explored in detail in this part.

Part II is organized to allow the reader to examine how performance data is analyzed and applied to the design process. Like the first edition, this book adopts a performance-based approach to design. Additionally, loading charts and scaling factors are also presented. Continuing with the theme of practical implementation, Part II also summarizes current knowledge that is key to getting wetland projects built, including construction methods, cost information, and operation and maintenance (O&M) requirements.

This book has been updated to reflect the dramatic advances in wetland technology over the last 13 years. The authors of this second edition come from very different backgrounds, and work in different aspects of the treatment wetland field. By combining our knowledge and experience, we have endeavored to present a broad range of information regarding the science, hydrology, hydraulics, reactor theory, applied design, construction, cost, and O&M of treatment wetland systems.

The focus of this book is almost entirely upon constructed wetlands rather than the use of natural wetlands. Although there are definitely circumstances in which it is logical and legal to utilize existing wetlands, it is far more common, at this point in wetland history, that a treatment wetland will be built on an existing upland site. The principles of operation and performance forecasting are not different.

1.1 WETLAND CHARACTERISTICS

Wetlands are land areas that are wet during part or all of the year because of their location in the landscape. Historically, wetlands were called *swamps, marshes, bogs, fens*, or *sloughs*, depending on existing plant and water conditions, and on geographic setting. Wetlands are frequently transitional between uplands (terrestrial systems) and continuously or deeply flooded (aquatic) systems. Wetlands are also found at topographic lows (depressions) or in areas with high slopes and low permeability soils (seepage slopes). In other cases, wetlands may be found at topographic highs or between stream drainages when land is flat and poorly drained (sometimes termed *blanket bogs* or *pocosins* in North America). In all cases, the unifying principle is that wetlands are wet long enough to exclude plant species that cannot grow in saturated soils and to alter soil properties because of the chemical, physical, and biological changes that occur during flooding.

There exists a wealth of published information about general wetland science. The reader may consult any of several texts, prominently including:

Biogeochemistry of Wetlands: Science and Applications, K.R. Reddy and R. DeLaune, 2008. CRC Lewis Publishers; Boca Raton, Florida.

Wetland Ecology Principles and Conservation, P.A. Keddy, 2000. Cambridge University Press; Cambridge, United Kingdom.

Wetlands, W.J. Mitsch and J.G. Gosselink, 2007. Fourth Edition, Wiley Publishers; New York, 600 pp.

Wetland Plants: Biology and Ecology, J.K. Cronk and M.S. Fennessey (Eds.), 2001. Lewis Publishers; Boca Raton, Florida, 462 pp.

Wetland Soils, J.L. Richardson and M.J. Vepraskas (Eds.), 2001. Lewis Publishers; Boca Raton, Florida, 417 pp.

Additionally, there are many compilations of research results for specific regional situations. These are for diverse locations, including, for example:

- **Czech Republic:**
  Freshwater Wetlands and Their Sustainable Future: A Case Study of the Trebon Basin Biosphere Reserve, Czech Republic, J. Kvet, J. Jenik, and L. Soukupova (Eds.), 2002. Parthenon Publishing; New York, 495 pp.
- **Canada:**
  Prairie Wetland Ecology, H.R. Murkin, A.G. van der Valk, and W.R. Clark (Eds.), 2000. Iowa State University Press; Ames, Iowa, 413 pp.
- **Florida:**
  Phosphorus Biogeochemistry in Sub-Tropical Ecosystems, K.R. Reddy, G.A. O'Connor, and C.L. Schelske (Eds.), 1999. Lewis Publishers; Boca Raton, Florida, 707 pp.

When combined with periodicals focused on wetlands, such as the journal *Wetlands*, these form a formidable collection of scientific works that explore many facets of wetland behavior and character.

Wetlands have properties that make them unique among the major ecosystem groups on Earth. Ample water is important for most forms of biological productivity, and wetland plants are adapted to take advantage of this abundant supply of water while overcoming the periodic shortage of other essential chemical elements, such as oxygen. Because of this, wetlands are among the most biologically productive ecosystems on the planet (Figure 1.1). As such, they are frequently inhabited by jungle-like growths of plants and are home to a multitude of animals including mammals, birds, reptiles, amphibians, and fish that are uncommon in other ecosystems.

FIGURE 1.1 A wide variety of birds and animals use treatment wetlands. (Photo courtesy R. Knight.)

In addition, because wetlands have a higher rate of biological activity than most ecosystems, they can transform many of the common pollutants that occur in conventional wastewaters into harmless byproducts or essential nutrients that can be used for additional biological productivity. These transformations are accomplished by virtue of the wetland's land area, with its inherent natural environmental energies of sun, wind, soil, plants, and animals. These pollutant transformations can be obtained for the relatively low cost of earthwork, piping, pumping, and a few structures. Wetlands are one of the least expensive treatment systems to operate and maintain. Because of the natural environmental energies at work in a wetland treatment system, minimal fossil fuel energy and chemicals are typically necessary to meet treatment objectives (Table 1.1).

TABLE 1.1
Energy Requirements for Different Types of Wastewater Treatment Plants

System	Hydraulic Load (m^3/d)	Energy Utilization (kW·h/m^3)	Reference
Surface Flow Wetlands	—	<0.1	Brix (1999)
Subsurface Flow Wetlands	—	<0.1	Brix (1999)
Facultative Lagoon + Rapid Infiltration	3,786	0.11	Campbell and Ogden (1999)
Facultative Lagoon + Overland Flow	3,786	0.16	Crites *et al.* (2006)
Aerated Subsurface Flow Wetlands	5,500	0.16	Wallace *et al.* (2006)
Tidal Flow (Fill-and-Drain) Wetlands	1,000	0.18	Maciolek and Austin (2006)
Carrousel Oxidation Ditch	3,786	0.51	U.S. EPA (1996)
Trickling Filter + Nitrogen Removal	3,786	0.61	Crites *et al.* (2006)
Activated Sludge + Nitrification	3,786	0.76	Campbell and Ogden (1999)
Extended Aeration Package Plant	3,786	1.06	U.S. EPA (1996)
Sequencing Batch Reactor	303	1.13	U.S. EPA (1996)
Living Machine	3,786	1.51	U.S. EPA (1996)

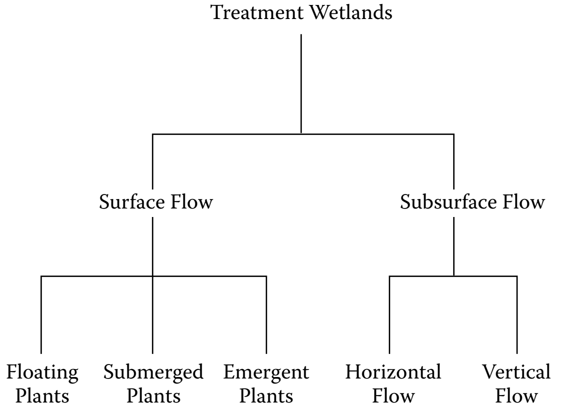


FIGURE 1.2 Treatment wetland types.

1.2 TYPES OF TREATMENT WETLANDS

Modern treatment wetlands are man-made systems that have been designed to emphasize specific characteristics of wetland ecosystems for improved treatment capacity. Treatment wetlands can be constructed in a variety of hydrologic modes. The basic types of constructed wetland systems are shown in Figure 1.2. At the current stage of technology development, three types of wetlands are in widespread use:

- Free water surface (FWS) wetlands have areas of open water and are similar in appearance to natural marshes.
- Horizontal subsurface flow (HSSF) wetlands, which typically employ a gravel bed planted with wetland vegetation. The water, kept below the surface of the bed, flows horizontally from the inlet to the outlet.
- Vertical flow (VF) wetlands distribute water across the surface of a sand or gravel bed planted with wetland vegetation. The water is treated as it percolates through the plant root zone. Biosolids dewatering wetlands can be thought of as a type of VF wetland system.

Each of these major categories employs variants of the layout, media, plants, and flow patterns. For example, FWS wetlands can be operated in intermittent flow, with a fill and drain mode, such as noted by Poach and Hunt (2007), even for continuous municipal discharges. The same operational strategy has also been advocated for HSSF systems (Behrends et al., 2001), and for VF systems (Sun *et al.*, 1999). VF wetlands may be operated in continuous downflow, as is the case for anaerobic mine water wetlands, or they may be operated with intermittent dosing, or they may be operated with intermittent or continuous sprinkling. Event-driven systems, such as stormwater wetlands, experience inflows that are erratic as well as intermittent.

FWS WETLANDS

These wetlands contain areas of open water, floating vegetation, and emergent plants, either by design or as an unavoidable consequence of the design configuration. Depending upon local regulations and soil conditions, berms, dikes, and liners can be used to control flow and infiltration. As the wastewater flows through the wetland, it is treated by the processes of sedimentation, filtration, oxidation, reduction, adsorption, and precipitation. The components in a typical FWS wetland are shown in Figure 1.3.

Because FWS constructed wetlands closely mimic natural wetlands, it should be no surprise that they attract a wide variety of wildlife, namely insects, mollusks, fish, amphibians, reptiles, birds, and mammals (NADB database, 1993; Kadlec and Knight, 1996). Because of the potential for human exposure to pathogens, FWS wetlands are rarely used for secondary treatment (U.S. EPA, 2000c). The most common application for FWS wetlands is for advanced treatment of effluent from secondary or tertiary treatment processes (e.g., lagoons, trickling filters, activated sludge systems, etc.). A typical application of a treatment train with a FWS wetland is shown in Figure 1.4.

FWS wetlands are suitable in all climates, including the far north. However, ice formation can hydraulically preclude winter operation, and the rates of some removal processes are lower for cold water temperatures, notably nitrogen conversion processes. When ice covers the open water, the transfer of oxygen from the atmosphere is reduced, decreasing oxygen-dependent treatment processes. Other processes, such as TSS removal, are more effective under the ice than in summer

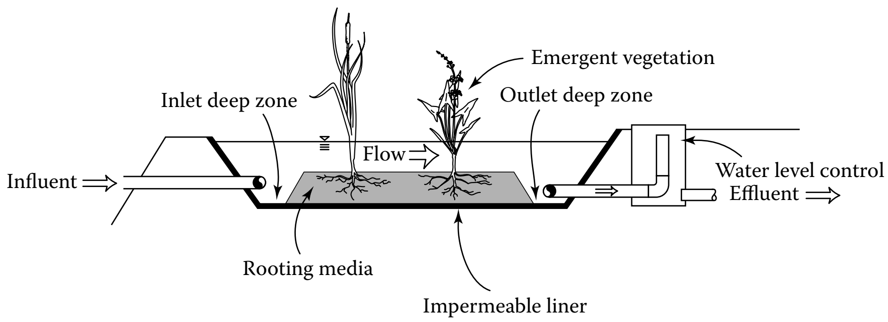


FIGURE 1.3 Basic elements of a FWS wetland.

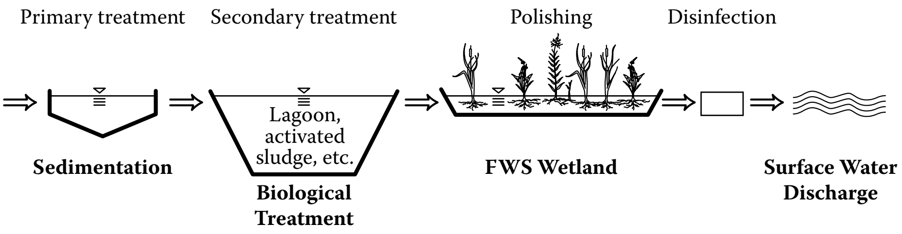


FIGURE 1.4 Typical application of a FWS wetland for municipal wastewater treatment. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

conditions. It is generally more efficient to store water during winter and treat it during the warm part of the year.

FWS wetlands are the nearly exclusive choice for the treatment of urban, agricultural, and industrial stormwaters, because of their ability to deal with pulse flows and changing water levels. They are a frequent choice for treatment of mine waters, and for groundwater remediation and leachate treatment.

FWS systems can provide significant ancillary benefits, primarily in the form of human uses and wildlife habitat. Treatment marshes are not inexpensive, but are usually capital cost-competitive with alternative technologies. Operating costs are typically quite low compared to alternatives.

HSSF Wetlands

HSSF wetlands consist of gravel or soil beds planted with wetland vegetation. They are typically designed to treat primary effluent prior to either soil dispersal or surface water discharge. The wastewater is intended to stay beneath the surface of the media and flows in and around the roots and rhizomes of the plants. Because the water is not exposed during the treatment process, the risk associated with human or wildlife exposure to pathogenic organisms is minimized. Properly operated HSSF wetlands do not provide suitable habitat for mosquitoes.

HSSF wetland systems are generally more expensive than FWS wetlands, although maintenance costs remain low compared to alternatives. They are commonly used for secondary treatment for single-family homes or small cluster systems (Wallace and Knight, 2006), or for small communities (Cooper *et al.*, 1996). However, there are many other applications to specialty wastewaters from industry. In general, HSSF wetlands have been utilized for smaller flow rates than FWS wetlands, probably because of cost and space considerations. HSSF wetlands are typically comprised of inlet piping, a clay or synthetic liner, filter media, emergent vegetation, berms, and outlet piping with water level control. A schematic of a conventional HSSF wetland for warm climates is depicted in Figure 1.5. A typical application of a HSSF wetland is shown in Figure 1.6.

These systems are capable of operation under colder conditions than FWS systems, because of the ability to insulate the top. A key operational consideration is the propensity for clogging of the media. HSSF wetlands do not provide the same opportunities for ancillary benefits that FWS systems do.

VF Wetlands

Several variations of VF wetlands exist. The most common type, used most often in Europe, employs surface flooding (pulse loading) of the bed in a single-pass configuration (Figure 1.7) (ÖNORM B 2505, 1997). Such systems are roughly analogous to the dosing scheme used in intermittent sand filters. VF wetlands in North America have been

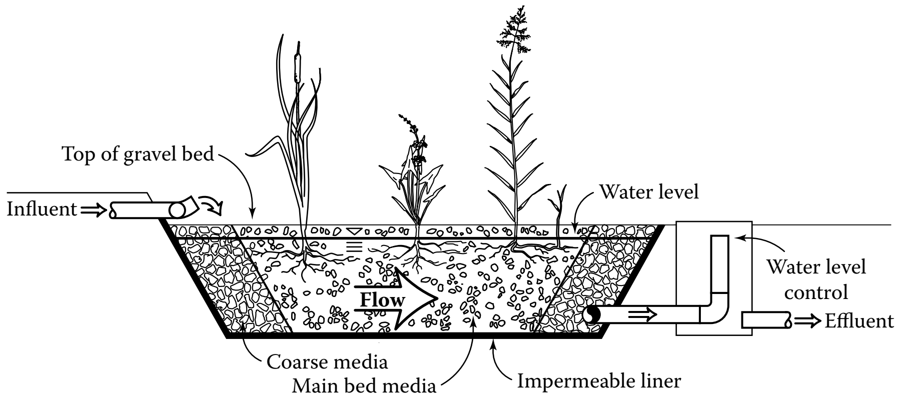


FIGURE 1.5 HSSF wetland schematic. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

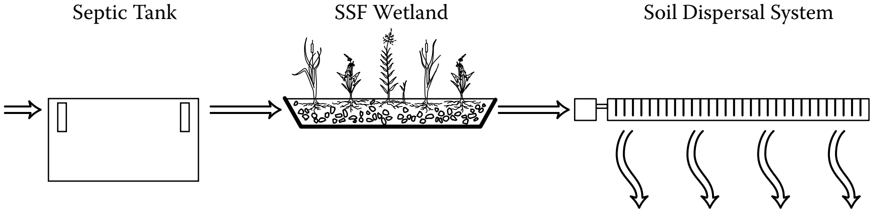


FIGURE 1.6 Application of a HSSF wetland to domestic wastewater treatment. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

designed as vegetated recirculating gravel filters (Lemon *et al.*, 1996). Upflow systems have been suggested to minimize oxygen transfer and promote reductive dehalogenation (Kassenga *et al.*, 2004), and fill-and-drain (tidal flow) systems have been implemented, mainly in North America, to treat high-strength wastes and to oxidize ammonia (Behrends *et al.*, 1996b; Austin and Lohan, 2005).

HSSF wetlands have a limited capacity to oxidize ammonia, because of limited oxygen transfer. VF wetlands were developed in Europe to provide higher levels of oxygen transfer, thus producing a nitrified effluent. The technology, initiated by Dr. Kathe Seidel in the early 1960s, became part of the Max Planck Institute Process (MPIP) (Brix, 1994d). These systems may be combined with HSSF or FWS wetlands to create nitrification-denitrification treatment trains (Figure 1.8; Cooper *et al.*, 1999).

The ability of VF wetlands to oxidize ammonia has resulted in their use in applications with higher ammonia than municipal or domestic wastewater. Landfill leachates and food processing wastewaters can have ammonia levels in the hundreds of milligrams per liter, and the key to reduction is the ability to nitrify. Successful VF wetlands therefore have formed part of the treatment process for those wastes (Burgoon *et al.*, 1999; Kadlec, 2003c).

Another variation of VF wetlands relies upon exactly the opposite process: the use of overlying water to block oxygen transport, in order to create anaerobic conditions in the bottom bed sediments. A surface water pool on top of organics and limestone creates downflow into a zone with reducing conditions that fosters appropriate sulfur chemistry to immobilize metals (Younger *et al.*, 2002).

Very concentrated wastewaters can be treated in VF systems. Unsettled raw sewage is added to VF wetlands in a French version of the technology (Molle *et al.*, 2005a), and sludge from activated sludge plants may be dewatered in VF systems (Nielsen, 2004).

Biosolids dewatering wetlands consist of an enclosed basin with alternating filter layers which trap organic biosolids on the surface of the wetland bed. Biosolids are applied to the surface of the wetland bed, and water percolates vertically down through the wetland bed primarily through mechanisms of unsaturated flow. Sludge dewatering systems target water removal and consolidation, rather than the elimination of dissolved constituents. Sludge dewatering beds consist of an enclosed basin with a sand layer underlain by drainage pipes. The sand bed is planted with emergent wetland plants (typically *Phragmites*), and fed throughout the year in intervals with up to 20 cm of stabilized sewage sludge per loading (Barjenbruch *et al.*, 2002). Solids content is typically 35–40% after dewatering (DeMaeseneer, 1997). Higher solids contents may be achieved, but this usually requires sacrificing the plants to drought stress (Nielsen, 1990). Freezing

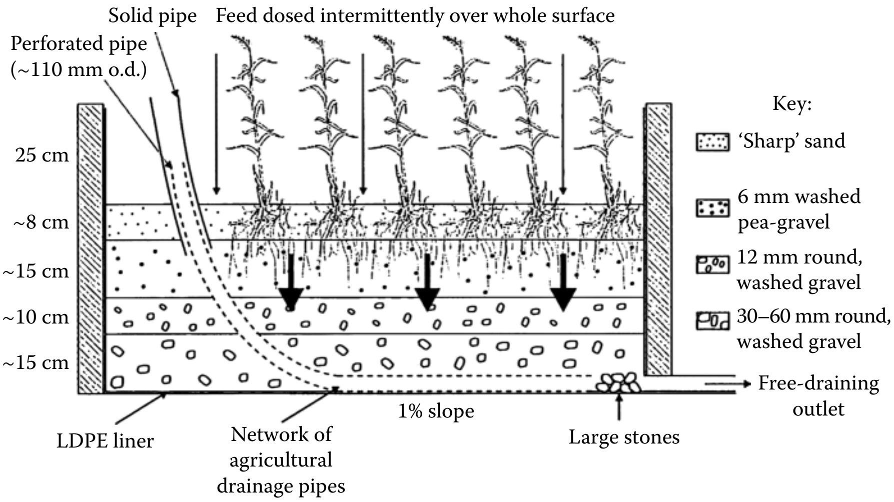


FIGURE 1.7 Typical arrangement of a VF constructed wetland. (From P.F. Cooper *et al.* (1996) *Reed Beds and Constructed Wetlands for Wastewater Treatment*. WRc Publications. Swindon, United Kingdom. Reprinted with permission.)

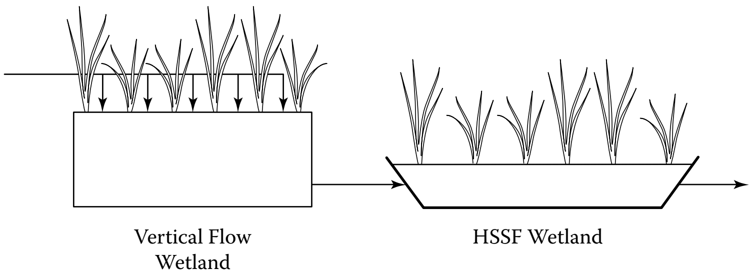


FIGURE 1.8 A hybrid wetland system (VF + HSSF).

conditions enhance performance since ice crystals lyse the cell walls of the bacteria in the biosolids (Reed *et al.*, 1995).

1.3 WETLANDS AS A TREATMENT TECHNOLOGY

Wastewater from human dwellings and activities has been a primary target of many treatment technologies, including treatment wetlands. Most of the early applications were for domestic and municipal wastewater, and that sector of the technology continues to grow at a rapid pace in many places. There are exceptions, such as Denmark, where some types of treatment wetlands have been implemented in essentially all locations where they make sense. However, there are a growing number of applications dealing with animal and industrial waters, urban and agricultural stormwaters, mine waters, groundwater remediation, and other applications. This diversification adds a second dimension to the growth of the use of treatment wetlands. It is useful to position the role of wetlands in the milieu of wastewater treatment in general.

Municipal Wastewater Treatment

Conventional municipal wastewater treatment is accomplished by physical, chemical, and biological processes. Many of these processes are general in nature and can function within a variety of treatment schemes. A review of these technologies is valuable for planning and designing a wetland treatment project for at least two reasons. First, pretreatment with conventional processes is usually advisable before discharge into a wetland because of the potential solids or oxygen demand overload that might create nuisance conditions within a wetland receiving raw or inadequately treated wastewaters. The wetland designer should be aware of the types of conventional processes that can be used to accomplish this pretreatment.

Secondly, a wetland treatment technology may not be the most cost-effective, environmentally sensitive, or technically reliable process for a given wastewater or project location. Conventional treatment technologies should be compared with wetlands and other land treatment technologies before final project planning and design is begun. A knowledge of conventional wastewater treatment methods, as well as the other natural land technologies, is essential to make a sound evaluation of the most appropriate treatment technology or combination of technologies for a given application. For detailed references on conventional municipal and domestic wastewater treatment technologies, see Metcalf and Eddy Inc. (1998) and Crites and Tchobanoglous (1998).

Primary Treatment

Primary treatment is considered "the first line of defense" in wastewater treatment (Water Environment Federation, 1988) because it sets the stage for the majority of biological treatment technologies that follow. Primary treatment consists of screening, grit removal, and primary sedimentation. Screening and grit removal may be referred to as *preliminary treatment* because they remove larger solids from the wastewater and the heavier mineral solids that might otherwise erode mechanical equipment downstream in the treatment facility.

Grit in raw wastewater primarily consists of inorganic and organic solids that enter the collection system and include materials such as sand, gravel, seeds, coffee grounds, and other minimally decomposable organic solids. Because grit is more settleable than more highly decomposable organic solids, it is removed in the front end of the treatment plant to protect mechanical equipment from abrasion and prevent sedimentation in pipelines and basins. Primary sedimentation is used to initially reduce the high concentration of total suspended solids (TSS) present in raw wastewater. Sedimentation is accomplished by creating quiescent flow conditions within a fairly deep (typically 3 to 5 m) pond or concrete vessel known as a primary clarifier. Settled solids are removed as sludge for further treatment, dewatering, and disposal, while the supernatant is removed via weirs to undergo additional treatment or discharge.

Secondary Treatment

Secondary treatment is the minimal level of municipal and industrial treatment that is required in the United States before discharge to most surface receiving waters. Secondary treatment requires a treatment level that will produce concentrations of five-day biochemical oxygen demand (BOD_5) and TSS less than 30 milligrams per liter (mg/L) and in addition, a minimum percent reduction of 85%.

Secondary treatment generally consists of the removal of additional wastewater solids and dissolved organic matter through microbial uptake and growth. Thus, secondary treatment is essentially a biological process in which bacteria and fungi are encouraged to grow in lagoons, mixed tanks, and ponds, or on fixed surfaces. The principal secondary treatment technologies are facultative ponds, aerated lagoons, aeration basins with solids recycling (activated sludge), trickling filters, and rotating biological contactors.

The activated sludge process is highly efficient at removing residual biochemical oxygen demand and suspended solids remaining in primary wastewater and can be adapted to also reduce ammonia nitrogen, total nitrogen, and phosphorus. A second group of secondary treatment technologies relies on attached growth of microbial populations to extract soluble carbon and nutrients from primary effluents. The trickling filter and rotating biological contactor technologies are two variations of this attached growth treatment process. Rotating biological contactors use circular plastic disks (media) mounted on a horizontal shaft and turning at about one to two rotations per minute (rpm) through a shallow wastewater-filled tank. Multi-unit rotating biological contactors are frequently designed in series, parallel, or both. Covers are typically provided over rotating biological contactors to minimize variation in the physical environment of the treatment process.

Advanced Treatment

Reductions of biochemical oxygen demand, total suspended solids, nitrogen, and phosphorus beyond those typically accomplished by secondary treatment are called tertiary or *advanced treatment*. Three forms of advanced wastewater treatment include nitrification, denitrification, and phosphorus removal. Nitrification can be accomplished in either suspended growth or in attached growth systems. Nitrification is an aerobic process in which bacteria oxidize ammonia to nitrate nitrogen. Standard activated sludge treatment can be modified to accomplish nitrification by increasing solids recycling (sludge age) and increasing the overall hydraulic residence time of the treatment system. Extended aeration activated sludge systems and oxidation ditch designs typically are capable of nitrification. Trickling filters and rotating biological contactor systems can also be designed for nitrification, especially through the use of multi-stage systems.

The total mass of nitrogen in the wastewater is not reduced by nitrification alone, but can be reduced by a second microbial transformation process called denitrification. In denitrification, nitrate nitrogen is microbially transformed into nitrogen gas, which is lost to the atmosphere. This process is anoxic, and occurs to a limited extent in conventional aerated treatment processes such as activated sludge or trickling filter units. Wastewater treatment systems can be designed for denitrification by including an anaerobic process after effluent nitrification. This process has been added in separate units as well as within single vessel units.

Normal microbial growth during secondary treatment results in a sludge with about 1.5–2% total phosphorus on a dry weight basis. Through sludge wasting, the total phosphorus content of the wastewater receiving secondary activated sludge treatment is reduced by about 10–25%. Biological phosphorus removal relies on a "luxury uptake" of phosphorus that occurs in microbial populations during growth in more vigorously aerated conditions. With higher uptake rates and increased sludge wastage, a higher percentage of the dissolved phosphorus can be removed from the wastewater. In addition, by sequencing through an anaerobic reactor before entry into the aerobic reactor, phosphorus content of the microbes is initially reduced, allowing a greater removal efficiency from solution in the second reactor.

Phosphorus removal from wastewaters is also frequently accomplished through several conventional chemical and physical processes. Chemical processes typically use aluminum (alum) or iron (ferric) salts to chemically precipitate dissolved phosphorus and remove it in a solid (sludge) form. Total phosphorus removal efficiency through chemical precipitation can exceed 90% in municipal effluents resulting in final total phosphorus concentrations lower than 0.5 mg/L. Physical phosphorus removal processes include ultrafiltration, reverse osmosis, and ion exchange. The first two physical processes rely on filtration of colloidal and dissolved phosphorus with a membrane, whereas ion exchange relies on the electrical attraction between ionized forms of phosphorus and specific ion exchange resins. All three of these physical processes require extensive pretreatment for suspended solids reduction and all generate a reject waste stream that may require additional chemical treatment for ultimate phosphorus removal.

Where Do Wetlands Fit?

Constructed wetlands may be used to provide some or all of the functions of secondary treatment and higher. Effluents that have undergone primary treatment may be further treated in constructed wetlands. At the present time, most such municipal wetlands are restricted to small communities and simple pretreatment systems. Common applications are:

- *Secondary treatment for small communities.* For example, Green and Upton (1992) analyzed the costs for HSSF systems in the United Kingdom, and concluded that they were the technology of choice for villages of up to 2,000 population.
- *Add-ons to aging or overloaded conventional secondary plants.* The wetland acts as a buffer to complete the treatment when there are upsets or extreme flow events that create bypass and concentration excursions in the conventional plant outflow.
- *Add-ons to lagoons.* The solids trapping properties of wetlands can compensate for the export of algal debris from facultative ponds, and provide further nutrient removal.

- *Tertiary and higher treatment of compliant secondary discharges.* Changing regulatory requirements can create the need for advanced treatment, which may be provided by constructed wetlands.

Domestic Wastewater Treatment

Although septic tanks and their accompanying drainfields have often served admirably in the partial treatment and disposal of wastewaters from single households or small dwelling clusters, that technology is limited in its capabilities and by site conditions. These systems do not control nitrogen, and in fact typically send oxidized nitrogen to groundwater. Clay soils, rock, and high groundwater tables may preclude effective infiltration. In such cases, the addition of a subsurface treatment wetland after the septic tank and preceding the drainfield can compensate for substandard infiltration conditions, and provide a greater level of nitrogen control. Treatment can be designed so that the wetland can discharge to surface waters, which is a frequent choice in Europe and developing countries.

Animal Wastewater Treatment

Constructed treatment wetlands are compatible with typical farm and ranch operations. Types of livestock wastewater being treated by constructed wetlands include dairy manure and milkhouse wash water, runoff from concentrated cattle-feeding operations, poultry manure, swine manure, and catfish pond water. The Livestock Wastewater Treatment Database (LWDB), created in 1998, included 68 sites with a total of 135 separate systems in North America (Knight *et al.*, 2000). The large majority of these were FWS wetlands. The strength of wastewater is higher than for municipal applications, with BOD, TSS, and ammonia often above 100 mg/L. Consequently, in contrast to other FWS applications, the treatment level may be characterized as primary.

Minewater Treatment

In the 1980s, a large number of treatment wetlands were built to treat acid mine drainage in the United States (Wieder, 1989). Constructed wetlands were in use at more than 300 sites in the United States in 1989, to increase the pH and reduce concentrations of iron and/or manganese at coal mine sites (Kleinman and Hedin, 1989). Conventional treatment of leachates at these sites would include surface grading and recontouring to reduce or divert flows, and chemical buffering and precipitation to improve water quality. Because these processes have relatively high capital and lifecycle costs, there was considerable interest in developing more cost-effective alternatives, and constructed wetlands were a logical choice. Methods of design were rudimentary, and remain so. Recent interest has grown to include metal mine wastewaters and tailings pile leachates (Younger, 2000). Applications include copper, gold, lead, and zinc mines.

Industrial Wastewater Treatment

A group of industries, characterized by their involvement in food processing, produce wastewaters that are high in biodegradable organic and nitrogen content. These wastewaters are typically quite strong, and routinely undergo some form of preliminary treatment. However, the reduction of nutrients and organics to regulatory levels is increasingly being accomplished by constructed wetlands. Application areas now involve wetlands serving the potato, wine, olive oil, sugar, starch, alcohol, and meat processing industries.

To meet reduced effluent limitations, some pulp and paper mills are being required to provide treatment beyond the secondary level. One goal may be to further reduce BOD_5, TSS, nitrogen, phosphorus, color, chlorinated organics (such as adsorbable organic halides or dioxin), and whole effluent toxicity. Constructed and natural wetland treatment systems have been used at pulp and paper mills to provide advanced secondary or tertiary treatment (NCASI, 2004).

Constructed wetlands are also providing advanced secondary and tertiary treatment of process water and stormwater at a growing number of petroleum refineries (Knight *et al.*, 1997). Typical wastewater pollutants at petroleum refineries include BOD_5, COD, oil and grease, TSS, NH_4-N, phenolics, H_2S, trace organics, and heavy metals. Concentrations of many of these pollutants are reduced through source control and preliminary treatments such as sour water stripping, oxidation and neutralization of spent caustics, and cooling tower blowdown treatment. Constructed wetlands are in use to reduce remaining concentrations of these contaminants to advanced treatment levels.

Leachate and Remediation

Treatment and disposal of liquid leachates is one of the most difficult problems associated with the use of sanitary landfills for disposal of solid waste. Leachates are produced when rainfall and percolated groundwater combine with inorganic and organic degraded waste. The highly variable nature of solid waste, differences in age and decomposition, and the diversity of chemical and biological reactions that take place in landfills result in a wide range of chemical quality of leachates. In unlined landfills, leachates frequently discharge to groundwater or appear as surficial drainage around the base of the landfill. In modern lined landfills, leachates are collected from the lined cells and routed to treatment units, or are trucked off-site to existing treatment plants. The use of constructed wetlands to treat these landfill leachates is a rapidly developing technology, with both subsurface flow (SSF) wetlands and surface flow wetlands (Mulamoottil *et al.*, 1998).

Groundwaters have been contaminated at a very large number of old industrial sites. For instance, the use of chlorinated ethenes was extremely prevalent up to about 30 years ago, at which time their attendant health hazards were recognized, and use discontinued. However, the dense and partially water-soluble materials had already been dumped to groundwater for many decades, leaving a legacy of contaminated groundwater. At some few sites, it was observed that removal was being achieved by natural wetlands, via the mechanisms of biodegradation and volatilization. It was only a small step to construct wetlands for the same purpose, and multiple sites now use wetland technology. Other common groundwater contaminants include hydrocarbons such as benzene, toluene, and other fuel hydrocarbons; explosives; and nitrates.

Alternatives to wetlands are extremely costly by comparison. Usually, the target chemicals can be more completely removed, but only with the expense of multistep processes involving chemical engineering technology. Those competing processes have a strong tendency to create large operations and maintenance requirements, and the presence of skilled operators. Such alternatives are especially onerous for remediation that is anticipated to last for many decades. The passive wetland alternative, with low or nonexistent replacement costs, presents a much better lifecycle prospect.

Urban Stormwater Treatment

Stormwater concentrations and loads are cyclic due to periods of dryfall and deposition, followed by the first flush of runoff after rain, followed by exponential decreases in runoff constituent concentrations as storages rinse from the landscape, and finally dry conditions and deposition until the next storm event. Pollutant concentrations and loads generally range from low levels from undeveloped and park lands, to low density residential and commercial, to higher density residential and commercial, and finally to high density commercial and industrial land uses. The use of constructed wetlands, usually with accompanying ponds, is now a routine best management practice (BMP) for controlling the quality of runoff. In the United States, the implementation of wetland stormwater BMPs has been very uneven, with numerous and early applications on both the east and west coasts, but much later and fewer systems elsewhere.

In contrast to other applications, there is basically no pretreatment for urban stormwaters, if the forebay settling basin is considered part of the wetland. At most, there may be a debris screen to catch major floating objects. Expectations are also typically lower than for many other application areas, with moderately good TSS reductions, but much lesser reductions in dissolve constituents. Although most urban stormwater wetlands are small, and do not acquire data, it is clear that their numbers are quite large.

Field Runoff Treatment

Target contaminants from agricultural fields vary, depending upon the perceived threat to receiving ecosystems. The principal contaminants include suspended solids, nitrate, phosphorus, and agricultural chemicals, but normally not all at the same time. Runoff from row crops and pasture areas may be low or high in mineral solids, depending on farming practices, rainfall intensity, soil types, and topography. Nutrient concentrations and loads from row crops and pastures depend on fertilization practices. As for urban runoff, there is usually no pretreatment prior to the wetland.

There is a potential role for wetlands in reducing solids loading coming from especially row crops. Such "dirt traps" are remarkably effective even at very short detention times (small systems) (U.S. Department of Agriculture, 1991; Braskerud, 2001a). However, short detention does not suffice to reduce dissolved nutrients, because those removals rely on the biogeochemical cycle, which operates at a much slower pace. Constructed wetlands also have the ability to abate the pulses of some pesticides, but not all, that are exported from fields in modern agriculture (Rodgers and Dunn, 1992). Despite such successes, the implementation of end-of-field wetlands has proceeded at a slow pace.

Wetlands are the only economically feasible means of controlling phosphorus in runoff reaching the Florida Everglades. The biggest constructed wetlands in the world are in operation there, with an aggregate area in excess of 20,000 ha. However, nitrogen pollution is a concern for the marine environments, and therefore wetlands are receiving considerable attention in connection with protection and restoration of the Baltic Sea and the Gulf of Mexico (Arheimer and Wittgren, 1994; Hey, 2002).

1.4 HISTORICAL PERSPECTIVE

Natural wetlands have been used as convenient wastewater discharge sites for as long as sewage has been collected (at least 100 years in some locations). Examples of old wetland sites in North America include the Great Meadows natural wetland near the Concord River in Lexington, Massachusetts, which began receiving wastewater in 1912; the Brillion Marsh in Wisconsin that has received municipal wastewater discharges since 1923; the Dundas sewage treatment plant, which began discharging to the Cootes Paradise natural wetland near Hamilton, Ontario, in 1919; and a discharge to a natural cypress swamp from the city of Waldo, Florida, since 1939.

Wetlands constructed for the purpose of treating water have a much shorter history. The worldwide spread of this technology originated from research conducted at the Max Planck Institute in West Germany, starting in 1952 (Bastian and Hammer, 1993) and in the western hemisphere during the 1970s. Implementation of wetland technology has been accelerating around the world since 1985, primarily because treatment wetlands, while mechanically simple, are biologically complex systems capable of achieving high levels of treatment. Furthermore, treatment wetlands can be constructed using local materials and local labor, which is a major advantage in developing countries.

Table 1.2 presents an annotated chronology of some of the major conferences leading to the acceptance of the use of natural and constructed wetlands for water quality management. The table lists selected research efforts, full-scale project

TABLE 1.2
Treatment Wetland Technology Conferences

Year	Location	Title (Proceedings)
1976	Ann Arbor, Michigan	Freshwater Wetland and Sewage Effluent Disposal (Tilton *et al.*, 1976)
1978	Tallahassee, Florida	Environmental Quality Through Wetlands Utilization (Drew, 1978)
1978	Lake Buena Vista, Florida	Wetland Functions and Values (Greeson *et al.*, 1979)
1979	Higgins Lake, Michigan	Freshwater Wetland and Sanitary Wastewater Disposal (Sutherland and Kadlec, 1979)
1979	Davis, California	Aquaculture Systems for Wastewater Treatment (Bastian and Reed, 1979)
1981	St. Paul, Minnesota	Wetland Values and Management (Richardson, 1981)
1982	Amherst, Massachusetts	Ecological Considerations in Wetlands Treatment of Municipal Wastewaters (Godfrey *et al.*, 1985)
1986	Orlando, Florida	Aquatic Plants for Water Treatment and Resource Recovery (Reddy and Smith, 1987)
1988	Chattanooga, Tennessee	1st International Conference on Constructed Wetlands for Wastewater Treatment (Hammer, 1989)
1989	Tampa, Florida	Wetlands: Concerns and Successes (Fisk, 1989)
1990	Cambridge, United Kingdom	2nd International Conference on Constructed Wetlands for Water Pollution Control (Cooper and Findlater, 1990)
1991	Pensacola, Florida	Constructed Wetlands for Water Quality Improvement (Moshiri, 1993)
1992	Columbus, Ohio	INTECOL Wetlands Conference (Mitsch, 1994)
1992	Sydney, Australia	3rd International Conference on Wetland Systems for Water Pollution Control (Pilgrim, 1992)
1994	Guangzhou, China	4th International Conference on Wetland Systems for Water Pollution Control (Chuncai, 1994)
1994	Atlanta, Georgia	On-Site Wastewater Treatment; 7th Symposium on Individual and Small Community Sewage Systems (Collins, 1994)
1995	Třeboň, Czech Republic	Nutrient Cycling and Retention in Wetlands and Their Use for Wastewater Treatment (Vymazal, 1996)
1996	Vienna, Austria	5th International Conference on Wetland Systems for Water Pollution Control (IGWA, 1996)
1996	Niagara-on-the-Lake, Ontario	Constructed Wetlands in Cold Climates (Friends of Fort George, 1996)
1997	Romulus, Michigan	Constructed Wetlands for the Treatment of Landfill Leachates (Mulamoottil *et al.*, 1998)
1997	Třeboň, Czech Republic	Nutrient Cycling and Retention in Natural and Constructed Wetlands (Vymazal, 1999)
1998	Aguas de São Pedro, Brazil	6th International Conference on Wetland Systems for Water Pollution Control (Tauk-Tornisielo and Salati, 1998)
1998	Orlando, Florida	On-Site Wastewater Treatment; 8th Symposium on Individual and Small Community Sewage Systems (Sievers, 1998)
1999	Salt Lake City, Utah	Wetlands and Remediation (Means and Hinchee, 2000)
1999	Třeboň, Czech Republic	Transformations of Nutrients in Natural and Constructed Wetlands (Vymazal, 2001)
1999	Baltimore, Maryland	Wetlands for Wastewater Recycling
1999	Tartu, Estonia	Constructed Wetlands for Wastewater Treatment in Cold Climates (Mander and Jenssen, 2002)
2000	Quebec, Canada	INTECOL Wetlands Conference (Pries, 2002)
2000	Orlando, Florida	7th International Conference on Wetland Systems for Water Pollution Control (Reddy and Kadlec, 2000)
2001	Burlington, Vermont	Wetlands and Remediation II (Nehring and Brauning, 2002)
2001	Třeboň, Czech Republic	Wetlands: Nutrients, Metals, and Mass Cycling (Vymazal, 2003)
2001	Fort Worth, Texas	On-Site Wastewater Treatment: 9th Symposium on Individual and Small Community Sewage Systems (Mancl, 2001)
2002	Dar es Salaam, Tanzania	8th International Conference on Wetland Systems for Water Pollution Control (Mbwette, 2002)
2003	Borová Lada, Czech Republic	Natural and Constructed Wetlands: Nutrients, Metals, and Management (Vymazal, 2005)
2003	Tartu, Estonia	Constructed and Riverine Wetlands for Optimal Control of Wastewater at Catchment Scale (Mander, 2003)
2003	Lisbon, Portugal	The Use of Aquatic Macrophytes for Wastewater Treatment in Constructed Wetlands (Dias and Vymazal, 2003)
2004	Wexford, Ireland	Nutrient Management in Agricultural Watersheds: A Wetlands Solution (Dunne *et al.*, 2005)
2004	Avignon, France	9th International Conference on Wetland Systems for Water Pollution Control (Liénard, 2004)
2005	Ghent, Belgium	1st Wetland Pollutant Dynamics and Control (WETPOL) (Tack *et al.*, 2005)
2006	Třeboň, Czech Republic	Wastewater Treatment, Plant Dynamics, and Management in Constructed and Natural Wetlands (Vymazal, 2008)
2006	Lisbon, Portugal	10th International Conference on Wetland Systems for Water Pollution Control (Dias and Vymazal, 2006)
2007	Padua, Italy	Multi-Functions of Wetland Systems (Borin and Bacelle, 2007)
2007	Tartu, Estonia	2nd Wetland Pollutant Dynamics and Control (WETPOL) (Mander *et al.*, 2007)

initiation dates, and key technical conferences at which the use of wetlands for water quality control was a featured topic.

In the early years of the technology development, it was possible and desirable to identify the numbers of systems, along with their characteristics. The benefit was the ability to document that constructed wetlands were being used in considerable numbers, and therefore were to be accorded some measure of recognition by the regulatory agencies and by the cadre of consulting engineers. For example, the North American treatment wetland database effort (NADB, Knight *et al.*, 1992) catalogued information on 127 treatment wetland systems, which was a springboard to data analyses that advanced the technology. However, growth has been exponential, and by 2005, a volunteer response survey produced a total of 497 small-scale constructed wetlands (flow less than 2,000 m^3/d) (Wallace and Knight, 2006). Enumeration is no longer a fruitful exercise, because there is no longer a need to demonstrate weight of numbers. It suffices to recognize that there are now many thousands of treatment wetlands across the globe. At this stage of technology development, attention is better directed to those systems that have, or are, producing data that enables optimization.

Development of Treatment Wetlands in North America

Wetland technology progress followed two parallel paths. The first, based on the recognition of the value of natural wetlands in water quality improvement, consisted of studies of projects that intentionally discharged wastewaters to existing wetlands. The second, the implementation of constructed wetlands, both FWS and HSSF, was initiated a few years later.

Natural Wetlands

Between 1967 and 1972, Howard T. Odum and A.C. Chestnut of the University of North Carolina began a five-year study of using coastal lagoons (with marsh wetland littoral vegetation) for recycling and reuse of municipal wastewaters (Odum, 1985). The studies included an examination of a natural *Spartina* salt marsh ecosystem that was receiving a discharge of secondarily treated wastewater (Camp *et al.*, 1971; Marshall, 1971; McMahan *et al.*, 1972; Stiven and Hunter, 1976).

In 1972, the University of Florida began a research effort directed at assessing the effectiveness of natural cypress wetlands for municipal wastewater recycling. From March 1974 until September 1977, secondarily treated municipal wastewater from a trailer park north of Gainesville was discharged to two isolated cypress wetlands (domes), and two control wetlands were also monitored. Research studies measured nearly all aspects of the physical, chemical, and biological processes occurring in the wastewater and control cypress domes (Ewel and Odum, 1984).

Concurrently, Robert Kadlec and coworkers at the University of Michigan began the first in-depth study of using engineered wetlands for wastewater treatment in a cold climate region. Work at the Porter Ranch peat-based wetland (peatland) located near the community of Houghton Lake, Michigan, began in 1971, with two years of discharges to 32 mesocosm plots in the peatland. A 360 m^3/d facility was operated with seasonal discharges for the next three years under the direction of university personnel. A full-scale system was initiated in 1978, and continues with a 2006 flow of 6,032 m^3/d during the summer to the Porter Ranch peatland (Figure 1.9). This system continues to operate today, and information from the Houghton Lake Natural Peatland Treatment System represents the longest data set on this aspect of the technology.

The public was not prepared to wait for results from these and other research efforts underway in the 1970s. In 1972, the city of Bellaire, Michigan, began discharging stabilized municipal wastewater to a 16-ha forested wetland (Kadlec, 1983). Although research was conducted on this system, the wetland was the primary means of effluent disposal for the city.

FIGURE 1.9 The Houghton Lake, Michigan, system utilizes a preexisting peatland for seasonal discharges.

The Reedy Creek Wetland Treatment System was implemented at Walt Disney World, Orlando, Florida, in 1977 (Knight *et al.*, 1987). The Reedy Creek system (Figure 1.10) used two wetlands, one with about 34 ha of natural mixed cypress/hardwood forested swamp, and the second with about 0.2 ha of constructed marsh and 5.6 ha of natural swamp forest to provide advanced wastewater treatment between 1977 and 1991 for monthly average flows as high as 22,700 m^3/d (7.2 cm/d). Flow to this wetland was discontinued in 1991 when a zero surface discharge option was implemented through landscape irrigation and groundwater recharge.

FIGURE 1.10 The Reedy Creek, Florida, system utilizes a preexisting forested wetland for wastewater polishing.

With wetlands being protected by federal law, the use of natural wetlands for wastewater treatment became limited at the end of the 20th century. Hammer and Bastian (1989) during the conference on constructed wetlands in Chattanooga, Tennessee, stated:

> Although some natural wetlands have been effectively used for water quality improvement, we do not wish to encourage additional use. We have recently become aware that natural wetlands are valuable resources that must not be wasted. Much remains to be learned about their many values and functions and the long-term consequences of wetland destruction. However, enough is known to conclude that it is not worth risking the unnecessary loss of any remaining natural wetlands without a better understanding of their important role in biological productivity, fish and wildlife habitat, flood protection, groundwater discharge, base flow stabilization of rivers, and water quality improvement. On the other hand, constructed wetlands may provide a relatively simple and inexpensive solution for controlling many water pollution problems without detrimentally affecting our natural wetlands resources. Although all of the processes are not well understood, constructed wetlands are capable of moderating, removing, or transforming a variety of water pollutants while also providing wildlife and recreational benefits commonly associated with natural wetland systems.

FWS Constructed Wetlands

In 1973, the Mt. View Sanitary District in Martinez, California, constructed about 8.5 ha of FWS wetland marshes for wildlife habitat and wastewater discharge (James and Bogart, 1989). Also in 1973, the first intentionally engineered, constructed wetland treatment pilot systems in North America were constructed at Brookhaven National Laboratory near Brookhaven, New York. These pilot treatment systems combined a marsh wetland with a pond and a meadow in series and were designated as the meadow/marsh/pond treatment system (Small, 1978). Industrial stormwaters and process waters were also applied to constructed pond/wetland systems as early as 1975 at Amoco Oil Company's Mandan Refinery in North Dakota (Litchfield and Schatz, 1989) (Figure 1.11). In 1976, the communities of Pinetop and Lakeside, Arizona, and in 1977 Show Low, Arizona, created a series of constructed lake/wetland areas for effluent evaporative disposal and wildlife production (Wilhelm *et al.*, 1989).

FIGURE 1.11 The Mandan, North Dakota, FWS system serves to polish refinery wastewaters.

North America has a rich history of constructing large-scale FWS treatment wetlands over the last 20 years. Florida has a number of the largest constructed wetland treatment areas in the world, including the Lakeland and Orlando constructed wetlands, both of which were started in 1987. Each wetland has about 500 ha for advanced treatment of municipal wastewater. Six treatment wetlands, called stormwater treatment areas, totalling over 16,000 ha, have been built in south Florida. These were designed to achieve a goal of 50 μg/L effluent phosphorus, and the last was placed in operation in 2004. Over 30 km of levees, and a comparable length of canals, are used to enclose each of these constructed treatment wetlands, and convey water. The approximate cost, including supporting research, was $1 billion, of which about $800 million was for construction. These are to treat an estimated thirty-year annual average flow of 4,400,000 m^3/d, and remove approximately 80 metric tons of phosphorus per year.

However, FWS treatment wetlands are by no means restricted to warm climates. Pries (1994) documented 67 constructed wetlands in Canada, including some in the Northwest Territories. The North American Database v. 2.0 (1998) has information for a total of 257 sites, 352 systems, and 622 cells from treatment wetlands in North America, and it is known that it was incomplete at the time of issuance. Of these 257 sites, 161 treat municipal wastewater, 10 receive industrial effluents, 68 receive livestock wastewaters, and 16 receive other wastewater types including stormwaters. Of the systems described in NADB v. 2.0, 270 are surface flow, 53 are HSSF, and 8 are hybrids of these two designs.

HSSF Constructed Wetlands

HSSF wetlands have gained widespread acceptance in North America as well. The first systems were built in 1972 near Seymour, Wisconsin, and researched through 1975 (Spangler *et al.*, 1976b). The researchers concluded that:

> Emergent vegetation has been used to treat wastewater biologically to a degree of purity which suggests that continued research could lead to widespread applicability of the process.

It took about ten years for the concept to develop, and by 1990, 98 HSSF systems were identified in the United States (Reed, 1990; 1991). Most of these (80) were in the southern states. The mean flow to these was 1,250 m^3/d, and the mean area was 3,400 m^2. Many of these were not properly designed to produce SSF, and operated in the flooded mode (Figure 1.12). In the 1990s, research was conducted at two primary HSSF sites in the United States: Baxter, Tennessee (George *et al.*, 1998), and Minoa, New York (Theis and Young, 2000). Results from those studies complemented the input–output data from other operating wetlands.

FIGURE 1.12 The Denham Springs, Louisiana, "HSSF" system was a lagoon add-on that could not carry the flow below the media surface.

FIGURE 1.14 The Salem, Oregon, VF system just after planting. (Photo courtesy of the city of Salem.)

By 2005, an effort to collect volunteer data identified several hundred HSSF systems in the United States, but this survey missed the majority of the single-home systems, because they have no reporting requirements (Wallace and Knight, 2006). For instance, in the United States, over 4,000 single-home HSSF systems are estimated to be in the state of Kentucky alone (Thom *et al.*, 1998). These single-home systems are subject to prescriptive design specifications, and very few produce data that may be used to further the technology development (Figure 1.13).

VF Constructed Wetlands

Very few VF wetlands have been implemented in North America, and these remain an area of technology development. Pulse-loaded VF systems based on European criteria are gaining increasing acceptance in North America (Kadlec, 2003c) (Figure 1.14). However, sprinkled beds with rotation are also in use (Burgoon *et al.*, 1999). Recirculating designs (based on gravel filters), a form of vertical flow wetlands, have been implemented in Canada (Lemon *et al.*, 1996).

Biosolids wetlands have been implemented at a number of mechanical wastewater treatment plants for stabilization of waste activated sludge, although the technology is not widespread. The current number of biosolids wetlands in North American is not known. There is an increasing interest in the technology from wastewater treatment plant operators, because biosolid wetland systems remain the simplest method to meet federally-mandated pathogen reduction requirements. To date, the technology has been used mainly in cold climates (Figure 1.15), because it is believed freezing during the winter aids in dewatering.

Treatment Wetlands in Europe

Development of constructed wetlands in Europe started with the work of Käthe Seidel, who began experimenting with aquatic macrophytes for water quality improvement (Seidel, 1953). This work was expanded in the 1950s and 1960s for various waste streams, including phenol wastewaters (Seidel, 1965; 1966), dairy wastewaters (Seidel, 1976), and livestock wastewaters (Seidel, 1961). The system evolved into a series of vertical and HSSF filter beds, and became known most commonly as the Max Planck Institute Process (MPIP) (Brix, 1994a; 1994d). This system was the basis for the "hybrid" wetland systems that were revived at the end of the 20th century.

In the mid-1960s, Seidel began collaboration with Reinhold Kickuth from Göttingen University. This collaboration ended after a few years due to personal reasons. Kickuth went on

FIGURE 1.13 (A color version of this figure follows page 550) Single-home HSSF wetland in Comfort Lake, Minnesota.

FIGURE 1.15 Biosolids stabilization wetland cells at Pine River, Minnesota.

to develop a HSSF wetland process commonly known as the root zone method (RZM). RZM wetlands were constructed with a soil media (typically clay loam to sandy clay) and planted with *Phragmites* in the belief that the root systems of this plant would improve the hydraulic conductivity of the media (Kickuth and Könemann, 1987).

The two scientists and their respective schools became rivals, producing conflicting information that created confusion among wastewater engineers and regulatory authorities (Brix, 1994a). By the 1980s, most constructed wetlands in Germany were RZM systems, although examples of the MPIP system were constructed in St. Bohaire, France (Liénard *et al.*, 1990), and Oaklands Park, United Kingdom (Burka and Lawrence, 1990).

HSSF Constructed Wetlands

The first full-scale RZM wetland into operation in 1974 in Liebenburg-Othfresen, Germany, for treatment of municipal wastewater (Kickuth, 1977). The area of about 22 ha was originally used to dump waste material (silt, clay, and dross) derived from mining of iron ore. Kickuth's concept of using heavy cohesive soils with low hydraulic conductivity was related to the traditional understanding of soil treatment of sewage, based on the "sewage farming" experiences in the United Kingdom (Cooper and Boon, 1987; Hiley, 1994). However, the predicted increases in the hydraulic conductivity of the bed media from root and rhizome growth did not occur, resulting in overland flow across the surface of the bed (Börner *et al.*, 1998).

In 1983, German designs (based on the root zone method) were introduced in Denmark, where the technology was recognized as being favorable for small community wastewater treatment. By 1987 about 80 horizontal flow constructed wetlands had been built (Brix, 1987; 1998). Despite problems with bed clogging and associated overland (surface) flow, these soil-based systems provided effective treatment if a bed area in excess of 3–5 m^2/PE was used. In order to overcome the overland flow problems, later Danish systems were designed with very wide beds and a short flow path (Brix, 1998). However, flow distribution was a problem with these very wide beds and the wetland was subdivided into two or more separate cells that could be loaded separately in order to get better control on the distribution of water (Brix, 1998).

In 1985, following visits to existing German and Danish systems, the first two HSSF constructed wetlands were built in the United Kingdom (where they are commonly called reed bed treatment systems). By the end of 1986, more than 20 systems had been designed (Cooper and Boon, 1987) (Figure 1.16). At the present time, there are over 1,000 systems in the database of the Constructed Wetland Association of the United Kingdom.

One major design change that was implemented in the United Kingdom was to switch to the use of coarser bed media (gravel) in order to maintain SSF within the wetland bed.

FIGURE 1.16 The HSSF reedbed at Acle, United Kingdom.

In the late 1980s, the first horizontal flow constructed wetlands were built in many European countries. By the 1990s, the technology had become a preferred method for wastewater treatment for small villages and other decentralized wastewater applications (Vymazal *et al.*, 1998).

The Mediterranean countries of Europe have developed large numbers of treatment wetlands, mainly over the past 15 years. Portugal documented 128 constructed wetlands in 2003 (Dias and Martins-Dias, 2003) which had grown to 176 by 2006 (Dias *et al.*, 2006), and they are growing in numbers in Spain, Italy, Greece, and Turkey.

Vertical Flow Constructed Wetlands

Constructed wetlands with VF date back to the original MPIP process developed by Seidel, where they were utilized as filtration beds in the first stage of the wetland treatment process (see Figures 15.9, 15.10).

The earliest full-scale VF wetlands were termed infiltration (or percolation) fields with VF through a soil or sand medium and with effluent discharged through underdrain pipes. This design was used to treat the wastewater from a recreation site in Lauwersoog, The Netherlands, in 1975 (Greiner and de Jong, 1984; Butijn and Greiner, 1985) (Figure 1.17). The system consisted of a preliminary settling/distribution ditch, four infiltration compartments, and an effluent ditch. Raw wastewater was discharged in the preliminary settling/distribution ditches. After settling, the water was intermittently fed to one of the VF wetland cells, which were alternately loaded and rested. This system has received intensive study (Rijs and Veenstra, 1990; Mueleman, 1999; Mueleman *et al.*, 2002).

VF constructed wetlands in Europe comprise a flat bed of graded gravel topped with sand that is planted with *Phragmites.* The beds are pulse-loaded with a large batch of water to temporarily flood the surface of the bed. Wastewater then percolates down through the bed via unsaturated flow. As the bed drains, air is drawn into the bed, reaerating the microbial biofilms. This pulse loading provides good oxygen transfer. As a result, VF wetland beds are known for their ability to nitrify (Cooper *et al.*, 1996).

FIGURE 1.17 Vertical flow wetland at Lauwersoog, The Netherlands.

VF constructed wetlands typically provide a good removal of organics and suspended solids, but these systems typically provide little denitrification. Consequently, removal of total nitrogen in these systems is limited.

VF constructed wetlands require less land (1–3 m^2/PE) as compared to horizontal flow systems (5–10 m^2/PE) but require more operation and maintenance. VF systems are very often used in Austria, Denmark, France, Germany, and the United Kingdom, especially for small sources of pollution. This wetland technology has been adopted in most European countries.

Hybrid Constructed Wetlands

Different types of constructed wetlands may be combined in order to achieve higher removal efficiency. These systems date back to the original MPIP system of Seidel. Currently, most hybrid systems employ combinations of horizontal and VF wetland cells. The most common configuration to date has been a VF stage followed by horizontal SSF wetland cells.

Over the last ten years, these types of vertical flow–horizontal flow systems were built in many European countries, such as Slovenia (Urbanc-Bercic and Bulc, 1994), Norway (Mæhlum and Stålnacke, 1999), Austria (Mitterer-Reichmann, 2002), and Ireland (O'Hogain, 2003). Hybrid systems are receiving more attention in most European countries because of more stringent requirements for ammonia removal.

An alternate hybrid wetland consisting of a horizontal flow bed followed by VF wetland cells has also been developed (Johansen and Brix, 1996). The large horizontal flow bed is placed first to remove organics and suspended solids and to provide denitrification. A pulse-loaded small VF bed is designed for further removal of organics and suspended solids and to nitrify ammonia to nitrate. A portion of the treated effluent is recirculated back to the influent in order to promote denitrification in the horizontal flow bed and improve total nitrogen removal (Brix *et al.*, 2003). Similar systems have been built in Poland at Sobiechy (Ciupa, 1996) and in Nepal at Dhulikhel in collaboration with Austrian researchers (Laber *et al.*, 1999).

More recent hybrid constructed wetlands use multiple wetland types, including FWS wetlands. An example of this approach can be found at Kõo in Estonia; this system consists of two VF beds, followed by a horizontal flow bed and two FWS wetlands (Mander *et al.*, 2003). In Italy, hybrid constructed wetlands are being successfully used for treatment of concentrated winery wastewaters (Masi *et al.*, 2002). The system at Ornellaia, Italy, consists of two VF beds, followed by a horizontal flow bed and a FWS wetland. The system at Cecchi, Italy, consists of horizontal flow beds followed by a FWS wetland and a pond.

FWS Constructed Wetlands

The IJsselmeer Polders Development Authority in Flevoland, The Netherlands, constructed the first European FWS wetland in 1967 (Veenstra, 1998). The wetland had a design depth of 0.4 m and the total area was 1 ha. A star-shape layout was chosen in order to obtain optimum utilization of the available area, however, this shape complicated macrophyte harvesting (de Jong, 1976). Therefore, longitudinal channels were added to facilitate mechanical biomass harvesting and system maintenance. The new wetland design included channels of 3 m wide and 200 m long (Figure 1.18), separated by

FIGURE 1.18 Free water surface (FWS) wetland at Elburg, The Netherlands.

parallel stretches of 3 m, resulting in an increase in land requirement from 5 m^2/PE for the star arrangement to 10 m^2/PE. The system exhibited a very good treatment effect and by the early 1970s, about 20 FWS wetlands of this type, called *planted sewage farms* (or Lelystad process farms), were in operation in The Netherlands (Greiner and de Jong, 1984; Veenstra, 1998).

In 1968, FWS-constructed wetlands were created in Hungary near Keszthely in order to preserve the water quality of Lake Balaton and to treat municipal wastewater (Lakatos, 1998). The constructed wetland was established in place of an existing natural wetland. The system originally consisted of six ponds 40–60 cm deep with a surface area of 10 ha. The ponds were fed with 8,000 m^3/d of mechanically pretreated wastewater. By 1985, the protection of Lake Balaton had grown to include the 1,800-ha Keszthely pond, which turned out to be a submerged aquatic vegetation (SAV) system (Clement *et al.*, 1998).

In contrast to North America, FWS-constructed wetland technology did not spread rapidly throughout Europe, and the main technology focus has been on HSSF and VF systems. However, FWS constructed wetlands are in operation in many European countries (e.g., Norway, Sweden, Denmark, Poland, Estonia, and Belgium). In Sweden, FWS systems have been constructed with nitrogen removal as a primary goal but other aims, such as biodiversity and irrigation, are also taken into consideration (Vymazal, 2006). Sometimes, the aim is to provide phosphorus polishing after chemical treatment and a buffer in case of treatment failure in the conventional treatment plant (Sunblad, 1998) (Figure 1.19). More than 2,350 ha of wetlands have been created in Sweden in the agricultural landscape between 1996 and 2002 in Denmark about 3,200 ha have been created prior to 2004 (Vymazal, 2006).

Biosolids Wetlands

The concept of vertical flow wetlands to remove organic matter extends back to the original system of Seidel (Seidel, 1965), but is also used in a modern context for VF wetlands in France, which are typically designed to accumulate biosolids

FIGURE 1.19 The treatment wetland at Oxelösund, Sweden.

FIGURE 1.20 Biosolids mineralization reedbeds at Skovby, Denmark. The bed in the foreground is newly excavated and replanted, and is receiving initial doses of biosolids. The bed in the left background is a mature stand of *Phragmites* that has been receiving biosolids for several years.

associated with raw sewage (Boutin *et al.*, 2002; Chazarenc and Merlin, 2004; Molle *et al.*, 2004a). Wetland beds designed specifically for biosolids dewatering have been most extensively developed in Denmark, where over 110 systems have been constructed since 1988 (Nielsen, 2006) (Figure 1.20). The largest current system is in Kolding, Denmark (123,000 PE). Use of wetlands for stabilization of organic biosolids is expanding throughout Europe (DeMaeseneer, 1997; Barjenbruch *et al.*, 2002; Obarska-Pempkowiak and Sobocinski, 2002; Lesavre and Iwema, 2002).

Treatment Wetlands in Australia, New Zealand, Africa, Asia, and South America

Australia

Aquatic macrophytes in Australia were initially evaluated for water quality improvement in the 1970s (Mitchell, 1976). In the 1980s, pilot-scale HSSF wetlands were evaluated for the treatment of piggery wastes and abattoir wastewater (Finlayson and Chick, 1983; Finlayson *et al.*, 1987). Extensive pilot-scale experiments, for both HSSF and FWS systems, were carried out at University of Western Sydney (Bavor *et al.*, 1987).

In 1992, the Cooperative Research Center for Constructed Wetlands was established, and several research projects were conducted on both FWS (e.g., the Byron Bay, New South Wales, full-scale system) and SSF wetlands (e.g., the Coff's Harbor, New South Wales, full-scale system). In the early 1990s, nine pilot wetlands were established in Queensland, of which eight were FWS (Greenway and Woolley, 1999; 2001).

Treatment wetlands for industrial wastewaters have been implemented; for instance, Noller *et al.* (1994) lists results from seven mine water wetlands in northern Australia, and oil refinery waters have also been treated (Simi, 2000).

Single-home HSSF wetlands have been extensively studied by Davison *et al.* (2001), but such domestic applications are still localized. In contrast, the application of stormwater treatment wetlands has proceeded with a considerable growth in numbers, in part spurred by the research endeavors of Wong and coworkers (Wong *et al.*, 1999; Wong *et al.*, 2006).

New Zealand

Some of the earliest constructed wetlands in New Zealand were for treatment of meat processing waters (Van Oostrom and Cooper, 1990; Van Oostrom and Russell, 1994). Research at the National Institute for Water and Air (NIWA), under the direction of C.C. Tanner, produced many valuable insights into the performance of HSSF wetlands over the period from 1994 to the present. The development and growth of constructed wetland technology has been stimulated by low investment and operating costs, and the technology, to some extent, addresses the Maori cultural and spiritual values.

According to a survey carried out by Tanner *et al.* (2000), constructed wetlands had been adopted enthusiastically by many New Zealand communities as a cost-effective means of secondary and tertiary wastewater treatment. Out of 83 constructed wetlands for wastewater treatment, excluding those treating stormwaters and farm dairy wastes, FWS treatment wetlands were most common (45%), followed by SSF and hybrid systems (35% and 14%, respectively). The remaining systems were called "enhanced natural wetlands." The surface flow systems were much larger (average size 2.2 ha) than those with SSF (average size 0.4 ha).

At present, constructed wetlands in New Zealand are also used to treat agricultural waters. Dairy runoff and pasture runoff are the focal points of new applications of wetlands.

Africa

Since the mid-1980s, the concept of using constructed wetlands gained support in Southern Africa, and by 1990, there were approximately 30 systems either in operation or under construction (Wood and Hensman, 1989; Batchelor *et al.*, 1990). These were intended to serve a number of functions, including treating raw sewage and secondary domestic effluents, septic tank and oxidation pond effluents, stormwaters, agricultural and aquaculture wastes, and a variety of industrial and mining wastewaters.

In the late 1990s, wetlands were piloted in Egypt, at Alexandria and at Abbu Attwa, Ismailia (Butler *et al.*, 1990). Several systems were implemented in Morocco (Mandi *et al.*, 1998; Radoux *et al.*, 2003). A very extensive constructed wetland demonstration project, the Bar el Baqar drain, located on one of the branches of the Nile as it enters Mediterranean estuaries, concluded in late 2006. Constructed wetlands have become more popular in central Africa, and there are now many examples of all types of constructed wetlands treating municipal sewage as well as industrial wastewaters and mine drainage waters (Kivaisi, 2001; Kaseva *et al.*, 2002; Mbuligwe, 2005; Abira *et al.*, 2005; Bojcevska *et al.*, 2006). Biosolids wetlands have also been implemented in Cameroon (Noumsi *et al.*, 2006) for stabilization of fecal sludges from primary settling tanks.

FIGURE 1.21 The treatment wetland system at Bainikeng, China.

Asia

The traditional expertise of Asian farmers in recycling human and animal wastes through aquaculture provides a good basis for what we choose to call "engineered wetland treatment systems" (Abbasi, 1987). As early as 1969, Sinha and Sinha reported on the use of the water hyacinth to treat digested sugar factory wastes. However, the first information about the use of constructed wetlands with emergent vegetation appeared only in the early 1990s (Juwarkar *et al.*, 1992). During the IWA conference in China in 1994, many papers on both horizontal and VF constructed wetlands from Asia, and especially China, were presented (Figure 1.21). Difficulties in language and communication have likely impeded the transfer of Asian information to the western world. At present, many constructed wetlands with emergent vegetation are in operation in India, China, Korea, Taiwan, Japan, Nepal, Malaysia, and Thailand for various types of wastewater.

South America

Since 1980, research has been conducted in Brazil on the possibility of the use of water hyacinth ponds in combination with vertical upflow constructed wetlands planted with rice (Salati, 1987). However, other types of constructed wetlands with emergent macrophytes have been adopted (Dallas *et al.*, 2004). There are limited numbers of constructed wetlands with emergent vegetation in South America, but systems are in operation in Brazil, Peru, Colombia, Honduras, Ecuador, Uruguay, Argentina, and also in Central America (Platzer *et al.*, 2002; Masi *et al.*, 2006).

SUMMARY

Constructed wetlands are mechanically simple treatment systems that rely primarily on passive treatment processes. These treatment systems are very favorable for use in rural settings or areas of low population density because they are relatively low-maintenance (compared to other treatment alternatives) and can usually be constructed from local materials. From global perspective, treatment wetland systems are gaining popularity as the market for cost-effective wastewater management expands in both developed and developing countries. This is primarily because treatment wetlands are perceived as a cost-effective and environmentally conscious treatment technology. There is also a growing realization that urban expansion may be best served by satellite wastewater treatment systems, rather that the continued expansion of centralized plants.

Natural wetlands have been used as receptors for wastewater since ancient times. The 20th century brought about the development of man-made wetland systems that are designed to emphasize specific characteristics of the natural wetland environment, aiming to improve the overall treatment capacity of the system. From this, three types of engineered wetlands have evolved for use in small-scale applications: FWS, HSSF, and VF.

FWS wetlands are similar to natural wetlands in that they contain areas of open water, floating vegetation, and emergent vegetation. They offer habitat benefits similar to natural wetlands, and invariably attract a variety of wildlife. These wetlands typically are used to polish effluent from secondary treatment processes such as lagoons, trickling filters, or activated sludge systems. They are rarely used as a stand-alone secondary treatment process due to their size and buffer requirements. FWS systems are virtually always the choice for stormwater treatment, and for animal wastewater treatment.

HSSF wetlands differ from FWS wetlands in that the wastewater is kept belowground. These wetlands are comprised of a lined gravel or soil-based bed planted with emergent vegetation. The wastewater is treated as it flows through the gravel media and around the roots and rhizomes of the plants. Because the wastewater is not exposed during the treatment process, the risk of pathogen exposure is minimized. HSSF wetlands are typically used to treat primary effluent to secondary treatment standards.

Vertical flow (VF) wetlands have found their widest application in Europe where the design goal is to produce a nitrified effluent. Because these systems accumulate biosolids on the surface of the bed, they may be incompatible with North American regulatory standards, which typically prohibit the surface exposure of fecal material. Nevertheless, VF systems are being used more and more in a global context.

Biosolids dewatering wetlands, a version of VF systems, are gaining an increasing level of support from operators of traditional sewage treatment works due to their simplicity and low operation and maintenance (O&M) requirements. This is especially true in cold climates where freezing promotes dewatering of accumulated biosolids.

The information summarized in this book represents the efforts of hundreds of scientists and engineers over the past three decades. While a synthesis of this massive collection of information is necessary to carry the wetland treatment technology to a wider audience, it is still beneficial for the reader to refer to the original sources for more details and a regional perspective, and to examine the evolution of wetland engineering during this formative period. The reader is encouraged to examine the references cited throughout the text and provided at the end of this book.

2 Hydrology and Hydraulics

The success or failure of a treatment wetland is contingent upon creating and maintaining correct water depths and flows. In this chapter, the processes that add and subtract water from the wetland are discussed, together with the relationships between flow and depth. Internal water movement in wetlands is a related subject, which is critical to understanding of pollutant reductions.

The water status of a wetland defines its extent, and is the determinant of plant species composition in natural wetlands (Mitsch and Gosselink, 2000). Hydrologic conditions also influence the soils and nutrients, which in turn influence the character of the biota. Flow and storage volume determine the length of time that water spends in the wetland, and thus the opportunity for interactions between waterborne substances and the wetland ecosystem.

The ability to control water depths is critical to the operation of treatment wetlands. This operational flexibility is needed to maintain the hydraulic regime within the hydrologic needs of desired wetland plant species, and is also needed to avoid unintended operational consequences, such as inlet zone flooding of horizontal subsurface flow (HSSF) treatment wetlands. It is therefore necessary to understand the hydraulic factors that relate depth and flow rate, including vegetation density and aspect ratio. In free water surface (FWS) wetlands, this requires an understanding of stem drag effects on water surface profiles. For HSSF and vertical flow (VF) wetlands, there are additional issues concerning the bed media size, hydraulic conductivity, and clogging.

2.1 WETLAND HYDROLOGY

Water enters wetlands via streamflow, runoff, groundwater discharge and precipitation (Figure 2.1). These flows are extremely variable in most instances, and the variations are stochastic in character. Stormwater treatment wetlands generally possess this same suite of inflows. Treatment wetlands dealing with continuous sources of wastewater may have these same inputs, although streamflow and groundwater inputs are typically absent. The steady inflow associated with continuous source treatment wetlands represents an important distinguishing feature. A dominant steady inflow drives the ecosystem toward an ecological condition that is somewhat different from a stochastically driven system.

Wetlands lose water via streamflow, groundwater recharge, and evapotranspiration (Figure 2.1). Stormwater treatment wetlands also possess this suite of outflows. Continuous source treatment wetlands would normally be isolated from groundwater, and the majority of the water would leave via streamflow in most cases. Evapotranspiration (*ET*) occurs with strong diurnal and seasonal cycles, because it is driven by solar radiation, which undergoes such cycles. Thus, *ET* can be an important water loss on a periodic basis.

Wetland water storage is determined by the inflows and outflows together with the characteristics of the wetland basin. Depth and storage in natural wetlands are likely to be modulated by landscape features, such as the depth of an adjoining water body or the conveyance capacity of an outlet stream. Large variations in storage are therefore possible, in response to the high variability in the inflows and outflows. Indeed, some natural wetlands are wet only a small fraction of the year, and others may be dry for interim periods of several years. Such periods of dry-out have strong implications for the vegetative structure of the ecosystem. Constructed treatment wetlands, on the other hand, typically have some form of outlet water level control structure. Therefore, there is little or no variation in water level, except in stormwater treatment wetlands. Dry-out in treatment wetlands does not normally occur, and only the vegetation that can withstand continuous flooding will survive.

The important features of wetland hydrology from the standpoint of treatment efficiency are those that determine the duration of water–biota interactions, and the proximity of waterborne substances to the sites of biological and physical activity. There is a strong tendency in the wetland treatment literature to borrow the detention time concept from other aquatic systems, such as "conventional" wastewater treatment processes. In purely aquatic environments, reactive organisms are distributed throughout the water, and there is often a clear understanding of the flow paths through the vessel or pond. However, wetland ecosystems are more complex, and therefore require more descriptors.

Hydrologic Nomenclature

Literature terminology is somewhat ambiguous concerning hydrologic variables. The definitions used in this book are specified below. The notation and parent variables are illustrated in Figure 2.1.

Hydraulic Loading Rate

The hydraulic loading rate (HLR, or q) is defined as the rainfall equivalent of whatever flow is under consideration. It does not imply uniform physical distribution of water over the wetland surface. In FWS wetlands, the wetted area is

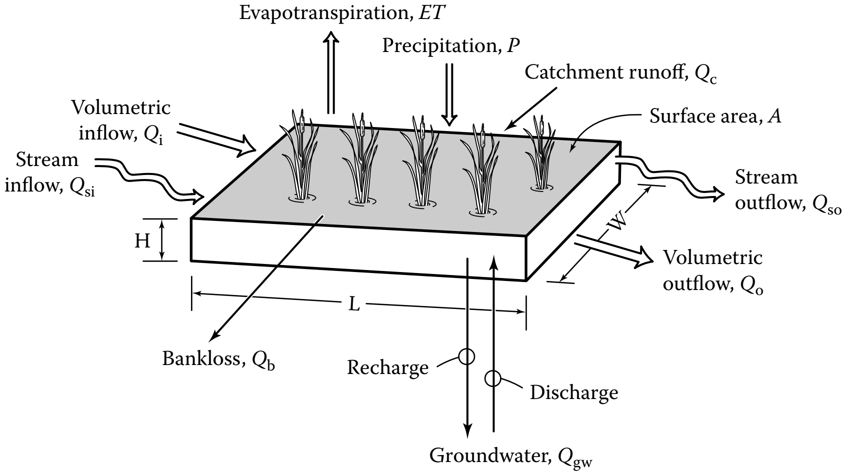


FIGURE 2.1 Components of the wetland water budget. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

usually known with good accuracy, because of berms or other confining features. The defining equation is:

$$q = \frac{Q}{A} \tag{2.1}$$

where
q = hydraulic loading rate (HLR), m/d
A = wetland area (wetted land area), m^2
Q = water flow rate, m^3/d

The definition is most often applied to the wastewater addition flow at the wetland inlet: $q_i = Q_i/A$. The subscript i, which denotes the inlet flow, is often omitted for simplicity.

Some wetlands are operated with intermittent feed, notably vertical flow wetlands. Under these circumstances, the term hydraulic loading rate refers to the time average flow rate. The loading rate during a feed portion of a cycle is *the instantaneous hydraulic loading rate*, which is also called the *hydraulic application rate*. Some wetlands are operated seasonally, for instance, during warm weather conditions in northern climates. Although these are in some sense intermittently fed, common usage is to refer to the loading rate during operation and not to average over the entire year. This means the instantaneous loading rate is used and not the annual average loading rate.

Mean Water Depth

Mean water depth is here denoted by the variable h. In FWS wetlands, the mean depth calculation requires a detailed survey of the wetland bottom topography, combined with a survey of the water surface elevation. The accuracy and precision must be better than normal, because of the small depths usually found in FWS wetlands. The two surveys combine to give the local depth:

$$h = H - G \tag{2.2}$$

where
G = local ground elevation, m
h = water depth, m
H = local water elevation, m

As-built surveys under dry conditions may not suffice for determination of ground levels, because of possible soil swelling and lift upon wetting. If the substrate is a peat or muck, there is not a well defined soil-water interface. Common practice in that event is to place the surveyor's staff "firmly" into the diffuse interface. Water surface surveys may be necessary in situations where head loss is incurred. This includes many HSSF wetlands, and some larger, densely vegetated FWS wetlands. Local water depth is then determined as the difference between two field measurements, and hence is subject to double inaccuracy.

The difficulties outlined above have prevented accurate mean depth determinations in many treatment wetlands. For example, detailed bathymetric surveys were conducted for a number of 0.2-ha FWS "test cells" in Florida (SFWMD, 2001) (Table 2.1). These were designed to be flat bottom wetlands, but proved to be quite irregular. The average coefficient of spatial variation in bottom elevations for seven of the ten cells was 39%. More importantly, there are errors ranging from –53 to +43% in the nominal volume of water in the wetlands. Errors of this magnitude have important consequences in the determination of nominal detention time.

HSSF wetlands typically have nonuniform hydraulic gradients due to clogging of the inlet region, as discussed further in this chapter. Therefore, the water depth may not be either flat or uniform in HSSF systems.

Wetland Water Volume and Nominal Detention Time

Free Water Surface Wetlands

For a FWS wetland, the nominal wetland water volume is defined as the volume enclosed by the upper water surface

TABLE 2.1
Bathymetry of Ten FWS Wetlands at the Everglades Nutrient Removal Project

Wetland Cell	Water Area (m^2)	Theoretical Depth (cm)	Measured Depth (cm)	Theoretical Volume (m^3)	Measured Volume (m^3)	Percent Difference
STC 1	2,251	60.0	54.9	1,255	1,140	10%
STC 2	2,296	15.0	12.4	341	280	22%
STC 4	2,474	30.5	21.3	754	528	43%
STC 9	2,534	32.6	45.4	826	1,151	–28%
STC 15	2,731	60.0	76.6	1,449	1,902	–24%
NTC 1	2,468	63.4	74.4	1,565	1,835	–15%
NTC 5	2,747	60.0	79.0	1,449	1,968	–26%
NTC 7	2,400	15.0	28.2	341	651	–48%
NTC 8	2,422	15.0	31.4	341	728	–53%
NTC 15	2,731	63.4	96.0	1,731	2,622	–34%

Note: STC = South Test Cell Site; NTC = North Test Cell Site.

and the bottom and sides of the impoundment. For a VF or HSSF wetland, it is that enclosed volume multiplied by the porosity of the media. *Actual wetland detention time* (τ) is defined as the wetland water volume involved in flow divided by the volumetric water flow:

$$\tau = \frac{V_{\text{active}}}{Q} = \frac{\varepsilon\, h\, A_{\text{active}}}{Q} \tag{2.3}$$

where

Q = flow rate, m^3/d
A_{active} = area of wetland containing water in active flow, m^2
h = wetland water depth, m
V_{active} = volume of wetland containing water in active flow, m^3
ε = porosity (fraction of volume occupied by water), dimensionless
τ = detention time, d

It is sometimes convenient to work with the nominal parameters of a given wetland. To that end, a *nominal detention time* (τ_n) is defined:

$$\tau_n = \frac{V_{\text{nominal}}}{Q} = \frac{(LWh)_{\text{nominal}}}{Q} \tag{2.4}$$

A very common alternative designation for nominal detention time is HRT. Equation 2.3 is a rather innocuous relation, but has no less than four difficulties, which have led to misunderstandings in the literature. First, there is ambiguity about the choice of the flow rate: Should it be inlet, or outlet, or an average? Differences in inlet and outlet flow rates are further discussed in this chapter.

Second, for FWS systems, some of the wetland volume is occupied by stems and litter, such that $\varepsilon \leq 1$. This quantity is difficult to measure, because of spatial heterogeneity, both vertical and horizontal. It is known to be approximately 0.95 for cattails in a northern environment (Kadlec, 1998), and for submerged aquatic vegetation (SAV) systems in the Everglades (Chimney, 2000), and for an emergent community (Lagrace *et al.*, 2000, as cited by U.S. EPA 1999).

Third, not all the water in a wetland may be involved in active flow. Stagnant pockets sometimes exist, particularly in complex geometries. As a result, $A_{\text{active}} \leq A = L{\cdot}W$. A *gross areal efficiency* may be defined as $\eta = A_{\text{active}}/A$. Fourth, the mean water depth (h) is difficult to determine with a satisfactory degree of accuracy, especially for large wetlands. That variability translates directly to a comparable uncertainty in the water depths, as noted in Table 2.1. These effects may be empirically lumped, and a volumetric efficiency (e_V) defined as:

$$e_V = \frac{V_{\text{active}}}{(LWh)_{\text{nominal}}} = \frac{\varepsilon\,\eta\, h}{h_{\text{nominal}}} \tag{2.5}$$

where

e_V = wetland volumetric efficiency, dimensionless
V_{active} = active wetland volume, m^3
ε = fraction of volume occupied by water, dimensionless
η = gross areal efficiency, dimensionless
h = water depth, m
h_{nominal} = nominal, water depth, m
LWh_{nominal} = nominal wetland volume, m^3

It is then clear that:

$$\tau = e_V \tau_n \tag{2.6}$$

Volumetric efficiency reflects ineffective volume within a wetland, compared to presumed nominal conditions. Portions of the nominal volume are blocked by submerged biomass (ε), bypassed (η), or do not exist because of poor bathymetry (h/h_{nominal}).

Confusion in nomenclature exists in the literature, where e_V is sometimes identified as *wetland porosity*. For dense emergent vegetation in FWS wetlands, this has presumptively been assigned a value in the range 0.65–0.75 (Reed *et al.*, 1995; Crites and Tchobanoglous, 1998; Water Environment Federation, 2001) (all of which use the symbol n in place of e_V). U.S. EPA (1999; 2000a) presumptively assigned the range 0.7–0.9 (both of which use the symbol ε in place of e_V).

It may be assumed that conservative tracer testing will provide a direct measure of the actual detention time in a wetland (Fogler, 1992; Levenspiel, 1995). Then, via Equation 2.6, there is a direct measure of e_V, although there is no knowledge gained about the three contributions to e_V by this process. At this point in the development of constructed wetland technology, there have been numerous such tracer tests. Summary results from 120 tests on 65 ponds and FWS wetlands present some insights (Table 2.2). First, the range of values for wetlands is indeed from 0.7 to over 0.9. But the range is even lower for basins devoid of vegetation, 0.55 to 0.9. That observation applies to the Stairs (1993) studies, which show empty basins with the same or lower e_V than identical geometries with plants (Table 2.2). This is a strong indication that the term *porosity* is a misnomer, because e_V is more strongly influenced by η and $h/h_{nominal}$.

Horizontal Subsurface Flow Wetlands

There is a very similar definition of e_V for HSSF systems:

$$e_V = \frac{V_{active}}{V_{nominal}} = \frac{\eta(\varepsilon V_{bed})}{V_{nominal}} \tag{2.7}$$

where

e_V = volumetric efficiency, dimensionless
ε = wetland bare media porosity, dimensionless
V_{bed} = actual wetland volume (water plus submerged media), m^3
$V_{nominal}$ = nominal wetland volume, m^3
η = gross volumetric efficiency, dimensionless

There is also uncertainty about the volumetric efficiency of subsurface flow wetlands. The mean porosity of a clean sand or gravel media is apt to be in the range 0.30–0.45 (Table 2.3). But, in an operational wetland, roots block some fraction of the pore space, as do accumulations of organic and mineral matter associated with treatment, which is accounted for by the gross areal efficiency, η. Roots block the upper horizons, and mineral matter preferentially settles to the bottom void spaces. Canister measurements of void fraction are not accurate, because of vessel wall effects and compaction problems. Attempts to measure water-filled void fraction by wetland draining have been thwarted by hold up of residual water. Wetland filling is an unexplored option for porosity determination. HSSF wetlands are often small enough to preclude significant errors in the determination of the bed or water depth, and thus it is expected that the ratio $V_{bed}/V_{nominal}$ is close to unity. It is therefore surprising to find a relatively wide spread in the measured values of e_V (Table 2.3). The range across the individual measurements was $0.15 < e_V < 1.38$. Interestingly, the mean across 22 HSSF wetlands is $e_V = 0.83$, which is virtually identical to that for FWS systems.

Spatial Flow Variation

There is obviously a possible ambiguity that results from the choice of the flow rate that is used in Equation 2.3 or 2.4.

TABLE 2.2
Hydraulic Characteristics of Ponds and Wetlands

Ponds (0.61–2.44 m deep)	Tests	Area (m^2)	L:W	Volumetric Efficiency, e_V	Reference
Three small scale	24	60–65	11.3	0.91	Lloyd *et al.* (2003)
One lab tank	3	75	6.75	0.74	Mangelson (1972)
Three pilot scale	3	1,148	4	0.55	Peña *et al.* (2000)
One pilot scale	5	1,323	3	0.74	Stairs (1993)
Ten dredge ponds	10	2,860–378,000	2.76	0.58	Thackson *et al.* (1987)
Mean			**5.56**	**0.70**	
Wetlands (0.3–0.8 m deep)	**Tests**	**Area (m^2)**	**L:W**	**Volumetric Efficiency, e_V**	**Reference**
Four pilots	18	1,323	3	0.78	Stairs (1993)
Six pilots	6	1,000–4,000	5.83	0.86	(1)
Sixteen pilots	24	1,200–13,400	3.95	0.69	(2)
Twenty-one pilots	27	2,700	3.30	0.96	(3)
Mean			**4.02**	**0.82**	

Sources: Unpublished data: (1) Champion Paper, (2) city of Phoenix, (3) Everglades Test Cells.

TABLE 2.3
Volumetric Efficiency of HSSF Wetlands

Study	Number of Tests	Wetlands	Porosity, ε	Volumetric Efficiency, e_V	Combined Effect, $\varepsilon \cdot e_V$
García (2003)	6	6	0.40	1.08	0.43
Chazarenc *et al.* (2003)	8	1	0.33	0.76	0.25
Rash and Liehr (1999)	5	2	0.41	0.28	0.12
Grismer *et al.* (2001)	2	2	0.36	1.02	0.37
Bavor *et al.* (1988)	3	3	0.33	0.93	0.31
Marsteiner (1997)	3	3	0.37	0.77	0.29
George *et al.* (1998)	5	5	0.36	1.08	0.40
Mean or Total	**32**	**22**	**0.37**	**0.83**	**0.30**

Wetlands routinely experience water gains (precipitation) and losses (evapotranspiration, seepage), so that outflows differ from inflows. If there is net gain, the water accelerates; if there is net loss, the water slows. A rigorously correct calculation procedure involves integration of transit times from inlet to outlet.

When there are local variations in total flow and water volume, the correct calculation procedure must involve integration of transit times from inlet to outlet. For steady flows, it may be shown that (Chazarenc *et al.*, 2003):

$$\tau_{an} = \tau_i \left(\frac{\ln(R)}{R-1} \right) \tag{2.8}$$

where
$R = Q_o/Q_i$, water recovery fraction, dimensionless
Q_i = inlet flow rate, m^3/d
Q_o = outlet flow rate, m^3/d
τ_{an} = actual nominal detention time, d
τ_i = inlet flow-based nominal detention time, d

In terms of detention time alone, moderate amounts of atmospheric gains or losses ($P - ET$) are not usually of great importance, although there is ambiguity in the choice of flow rate (Q). Some authors base the calculation on the average flow rate (inlet plus outlet ÷ 2). This approximation is good to within 4% as long as the water recovery fraction is $0.5 < R < 2.0$.

Velocities and Hydraulic Loading

The relation between nominal detention time and hydraulic loading rate is:

$$q = \frac{Q_i}{LW} = \frac{\varepsilon h}{\tau_n} \tag{2.9}$$

where
q = hydraulic loading rate, m/d
Q_i = inlet flow rate, m^3/d
L = wetland length, m
W = wetland width, m
ε = porosity of wetland bed media, dimensionless
h = water depth, m
τ_n = nominal hydraulic retention time, d

Thus, it is seen that hydraulic loading rate is inversely proportional to nominal detention time for a given wetland depth. Hydraulic loading rate therefore embodies the notion of contact duration, just as nominal detention time does.

The *actual water velocity* (v) is that which would be measured with a probe in the wetland—a spatial average. In terms of the notation used here:

$$v = \frac{Q}{\varepsilon h W} \tag{2.10}$$

where
v = actual water velocity, m/d
Q = flow rate, m^3/d
W = wetland width, m
ε = wetland bed porosity, dimensionless
h = water depth, m
εhW = open area perpendicular to flow, m^2

It is noted that there is large spatial and temporal variation in v, and hence individual spot measurements may be as much as a factor of ten different from the mean. Field investigations tend to have a bias towards high local measurements because probes do not easily find small pockets of stagnant water.

The *superficial water velocity* (u) is the empty wetland velocity—again, a spatial average. In terms of the notation used here:

$$u = \frac{Q}{hW} \tag{2.11}$$

where
u = superficial water velocity, m/d
Q = flow rate, m^3/d
W = wetland width, m
h = water depth, m
hW = total wetland area perpendicular to flow, m^2

For FWS wetlands, there is not much difference between u and v, because FWS porosity is nearly unity (typically around 0.95). However, there is a large difference for HSSF systems because of the porosity of the bed media (typically around 0.35–0.40). Superficial water velocity (u) is used in the technical literature on water flow and porous media, and care must be taken to avoid misuse of those literature results.

The relation between superficial and actual velocities is:

$$u = \varepsilon v \tag{2.12}$$

where
u = superficial water velocity, m/d
ε = wetland bed porosity, dimensionless
v = actual water velocity, m/d

Overall Water Mass Balances

Transfers of water to and from the wetland follow the same pattern for surface and subsurface flow wetlands (see Figure 2.1). In treatment wetlands, wastewater additions are normally the dominant flow, but under some circumstances, other transfers of water are also important. The dynamic overall water budget for a wetland is:

$$Q_i - Q_o + Q_c - Q_b - Q_{gw} + Q_{sm} + (P \times A) - (ET \times A) = \frac{dV}{dt} \tag{2.13}$$

where
A = wetland top surface area, m^2
ET = evapotranspiration rate, m/d
P = precipitation rate, m/d
Q_b = bank loss rate, m^3/d
Q_c = catchment runoff rate, m^3/d
Q_{gw} = infiltration to groundwater, m^3/d
Q_i = input wastewater flowrate, m^3/d
Q_o = output wastewater flowrate, m^3/d
Q_{sm} = snowmelt rate, m^3/d
t = time, d
V = water storage (volume) in wetland, m^3

Inflows and Outflows

Most moderate to large scale facilities will have input flow measurement; a smaller number of facilities will have the capability of independently measuring outflows as well as inflows. Due a lack of outlet flow measurements, the overall water budget Equation 2.13 is often used to calculate the estimated outflow rate. Usually, only rainfall is a significant addition, and only ET is a significant subtraction, to the inflow, simplifying the analysis. This calculation is most easily performed when there is no net change in storage.

The change in storage (ΔV) over an averaging period (Δt) can be a significant quantity compared to other terms in the water budget. For example, if the nominal detention time in the wetland is 10 days, then a 10% change in stored water represents one day's addition of wastewater. Because water depths in treatment wetlands are typically not large, changes of a few centimeters may be important over short averaging periods. If there is significant infiltration, there are two unknown outflows (Q_o and $Q_{gw} + Q_b$), and the water budget alone is not sufficient to determine either outflow by difference.

Rainfall

Rainfall amounts may be measured at or near the site for purposes of wetland design or monitoring. However, the gauging location must not be too far removed from the wetland, because some rain events are extremely localized.

For most design purposes, historical monthly average precipitation amounts suffice. These may be obtained from archival sources, such as Climatological Data, a monthly publication of the National Oceanic and Atmospheric Administration (NOAA), National Climatic Data Center, Asheville, North Carolina. In the United States, a very large array of climatological data products are available online at www.ncdc.noaa.gov/oa/climate/climateproducts.html. As an illustration of that service, the (free) normal precipitation map is shown in Figure 2.2.

The total catchment area for a wetland is likely to be just the area enclosed by the containing berms and roads; and that area is easily computed from site characteristics. Rainfall on the catchment area will, in part, reach the wetland basin by overland flow, in an amount equal to the runoff factor times the rainfall amount and the catchment area (Figure 2.3). A very short travel time results in this flow being additive to the rainfall:

$$Q_c = \Psi P A_c \tag{2.14}$$

where
Q_c = flow rate from contributing catchment area, m^3/d
A_c = catchment surface area, m^2 (does not include the net wetland area)
Ψ = catchment runoff coefficient, dimensionless (1.0 represents an impervious surface)
P = precipitation, m

For small and medium sized wetlands, the catchment area will typically be about 25% of the wetland area, as it is for the Benton, Kentucky, system, for example. About 20% of a site will be taken up by berms and access roads which may drain to the wetland. Runoff coefficients are high, because the berms are impermeable; a range of 0.8–1.0 might be typical. The combined result of impermeable berms, their necessary area, coupled with quick runoff, is an addition, of about 20–25% to direct rainfall on the bed.

Dynamic Rainfall Response

Many treatment wetland systems are fed a constant flow of wastewater. There is therefore a strong temptation amongst

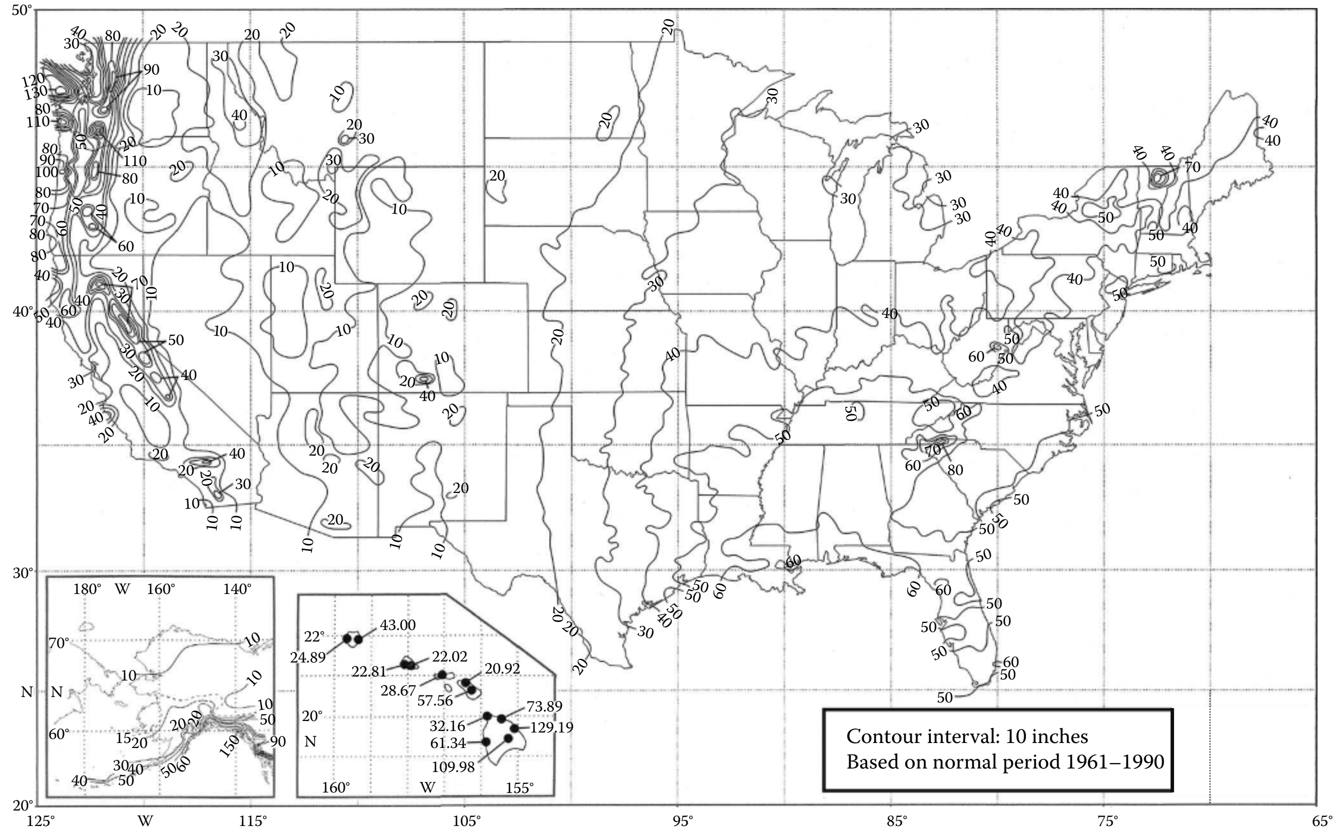


FIGURE 2.2 Normal precipitation map for the United States.

wetland designers to visualize a relatively constant set of system operating parameters—depths and outflows in particular. This is not the case in practice. There may be significant outflow response to rain events. A sudden rain event, such as a summer thunderstorm, will raise water levels in the wetland. The amount of the level change is magnified by catchment effects, and bed porosity in the case of HSSF systems. A relatively small 3-cm rain event can raise HSSF bed water levels by more than 10 cm. This often exceeds the available head space in the wetland bed. As a result, HSSF wetlands typically experience short-term flooding in response to large storm events and berm heights are usually designed to temporarily store a specified amount of rainfall (such as a 25-year, 24-hour storm event) above the HSSF bed. In any case, outflows from the system increase greatly as the rainwater flushes from the system.

As an illustration, consider Cell #3 at Benton, Kentucky, in September, 1990. Figure 2.4 shows a rain event of about 2 cm occurring at noon on September 10, 1990. The HSSF bed was subjected to a surplus loading of over 100% of the daily feed in a brief time period. The result was a sudden increase in outflow of about 300%, which subsequently tapered off to the original flow condition.

The implications for water quality are not inconsequential. In this example, samples taken during the ensuing day represent flows much greater than average. Water has been pushed through the bed, and exits on the order of one day early; and has been somewhat diluted. Velocity increases are great enough to move particulates that would otherwise remain anchored. Internal mixing patterns will blur the effects of the rain on water quality.

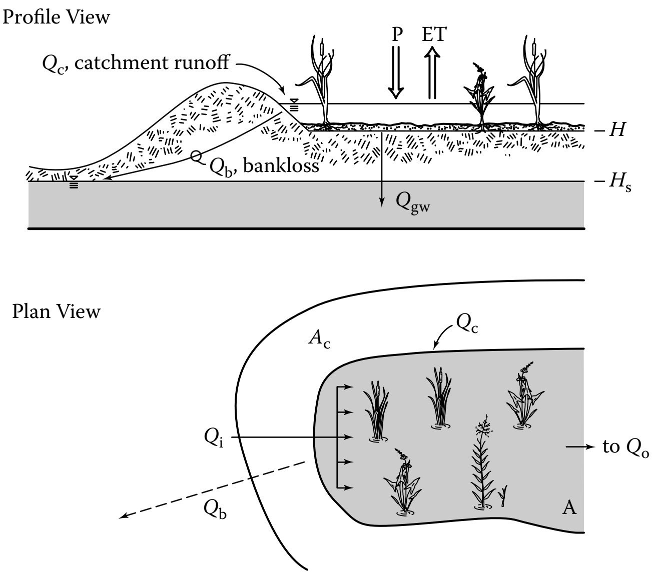


FIGURE 2.3 Water budget quantities. (Adapted from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

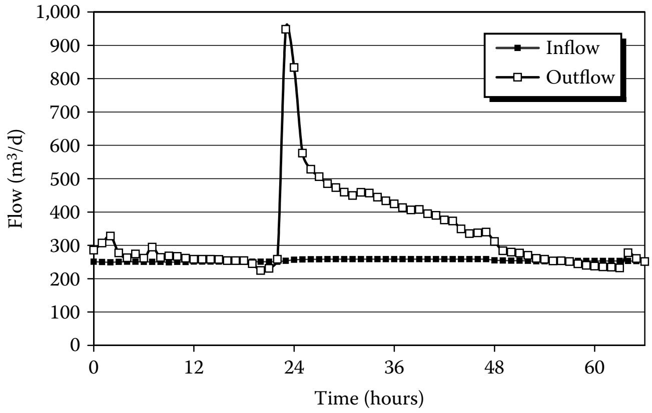


FIGURE 2.4 Flows into and out of Benton Cell #3 versus time during a rain event period during September 9–11, 1990. Flows were measured automatically via data loggers; the values were stored as hourly averages. The rain event totaled approximately 1.90 cm, or 278 m^3. (Data from TVA unpublished data; graph from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

Sampling intervals are not normally small enough to define these rapid fluctuations. For instance, weekly sampling of Benton Cell #3 would have missed all of the details of the rain event in the illustration above. It is therefore important to realize that compliance samples may give the appearance of having been drawn from a population of large variance, despite the fact that the variability is in large part due to deterministic responses to atmospheric phenomena.

Evapotranspiration

Water loss to the atmosphere occurs from open or subsurface water surfaces (evaporation), and through emergent plants (transpiration). This water loss is closely tied to wetland water temperature, and is discussed in detail in Chapter 4. Here the impacts of evapotranspiration (*ET*) on the wetland water budget are explored. At this juncture the two simplest estimators will be noted: Large FWS wetland *ET* is roughly equal to lake evaporation, which in turn is roughly equal to 80% of pan evaporation. Table 2.4 shows the distribution of monthly and annual lake evaporation in different regions of the United States.

Wetland treatment systems frequently operate with small hydraulic loading rates. For 100 surface flow wetlands in North America, 1.00 cm/d was the 40th percentile

TABLE 2.4
Lake Evaporation (in mm) at Various Geographic Locations in the United States

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Yuma, Arizona	99	117	165	203	249	292	340	328	272	203	155	114	2,540
Sacramento, California	20	36	64	91	127	180	226	218	180	122	66	30	1,372
Denver, Colorado	41	46	64	94	127	188	224	213	170	117	76	48	1,397
Miami, Florida	76	86	104	124	127	122	135	130	109	104	109	69	1,295
Macon, Georgia	43	56	79	109	130	157	160	147	132	107	71	46	1,245
Eastport, Maine	20	18	23	28	36	43	51	53	51	41	28	18	406
Minneapolis, Minnesota	8	10	23	43	81	112	152	147	117	76	33	10	813
Vicksburg, Mississippi	33	48	74	107	127	145	147	140	132	112	74	41	1,168
Kansas City, Missouri	23	28	43	79	112	155	203	198	152	114	64	25	1,194
Havre, Montana	13	13	28	64	114	155	208	211	142	84	38	18	1,092
North Platte, North Dakota	20	28	56	94	127	165	218	213	175	117	66	28	1,295
Roswell, New Mexico	53	81	124	173	211	249	239	211	175	140	89	64	1,803
Albany, New York	15	18	28	51	81	109	132	119	86	61	36	20	762
Bismarck, North Dakota	10	13	25	58	102	135	185	196	147	84	33	13	991
Columbus, Ohio	15	20	28	58	89	117	142	130	104	76	41	15	838
Oklahoma City, Oklahoma	38	48	79	119	140	198	259	272	224	160	89	51	1,676
Baker, Oregon	13	18	36	64	86	112	175	185	124	74	38	15	940
Columbia, South Carolina	41	61	81	114	137	160	168	152	140	112	76	48	1,295
Nashville, Tennessee	23	33	48	84	104	130	147	137	124	94	53	28	991
Galveston, Texas	23	33	41	66	104	142	157	155	145	117	69	33	1,092
San Antonio, Texas	56	79	114	142	165	213	239	239	193	147	94	61	1,753
Salt Lake City, Utah	20	25	51	89	130	201	269	264	185	99	51	25	1,397
Richmond, Virginia	33	43	56	89	104	127	142	124	104	81	61	38	991
Seattle, Washington	20	20	36	53	69	86	99	86	66	41	28	18	610
Milwaukee, Wisconsin	15	18	23	33	53	81	127	137	119	81	41	15	737

Source: From van der Leeden *et al.* (1990) *The Water Encyclopedia.* Second Edition, Lewis Publishers, Boca Raton, Florida.

in the early days of constructed wetland technology (NADB database, 1993). *ET* losses approach a daily average of 0.50 cm/d in summer in the southern United States; consequently, more than half the daily added water may be lost to *ET* under those circumstances. But *ET* follows a diurnal cycle, with a maximum during early afternoon, and a minimum in the late nighttime hours. Therefore, outflow can cease during the day during periods of high *ET*.

As a second example, Platzer and Netter (1992) report that the nominal detention time, based on inflow, for the subsurface flow wetland at See, Germany, was 20 days. There was a measured net loss of 70% of the water to evapotranspiration in summer. The actual nominal detention time, computed from Equation 2.8, is 34.4 days; the use of an average flow rate gives 30.8 days.

In addition to the consumptive use of water, which may be critical in water-poor regions, *ET* acts to concentrate contaminants remaining in the water. For instance, Platzer and Netter (1992) report that the wetland accomplished 88% ammonia removal on a mass basis. When coupled with the 70% water loss, the ammonia concentration reduction is only 60%.

In mild temperate climates, annual rainfall typically slightly exceeds annual *ET*, and there is little effect of atmospheric gains and losses over the course of a year. But most climatic regions have a dry season and a wet season, which vary depending upon geographical setting. As a consequence evapotranspiration losses may have a seasonally variable impact. For example, *ET* losses are important in northern systems that are operated seasonally. In northern North America, about 80% of the annual *ET* loss occurs in the six months of summer. Therefore, lightly loaded seasonal wetlands in cold, arid climates are strongly influenced by net atmospheric water loss. Examples include the Williams Pipeline HSSF system in Watertown, South Dakota (Wallace, 2001), which operates at zero discharge during the summer, the Roblin, Manitoba, FWS system, which operates at zero discharge two summers out of every three; and the Saginaw, Michigan, FWS system, which operates with 50% water loss (Kadlec, 2003c).

Dynamic ET Response

The diurnal cycle in *ET* can be reflected in water levels and flow rates under light loading conditions. HSSF Cell #3 at Benton, Kentucky, was operated in September 1990 at a hydraulic loading rate (HLR) of 1.7 cm/d, corresponding to a nominal detention time (HRT) of approximately 13 days. Evapotranspiration at this location and at this time of year was estimated to be about 0.5 cm/d. Consequently, *ET* forms a significant fraction of the hydraulic loading. Because *ET* is driven by solar radiation, it occurs on a diurnal cycle. The anticipated effect is a diurnal variation in the outflow from the bed, with amplitude mimicking the amplitude of the combined (feed plus *ET*) loading cycle. This was measured at Benton (Figure 2.5).

In such an instance, because the night outflow peak is nearly double the daytime minimum outflow, it would be desirable to use diurnal timed samples of the outflow, and to appropriately flow-weight them, for determination of water quality.

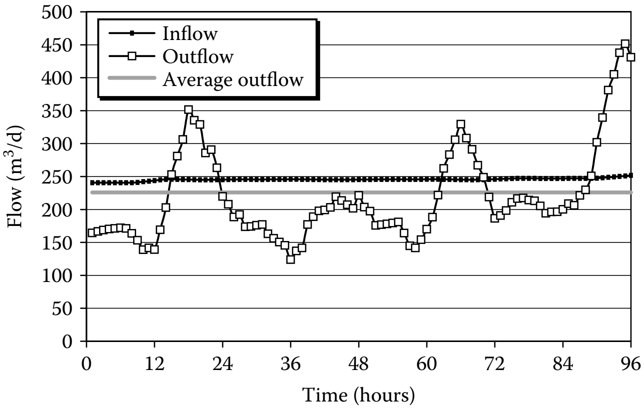


FIGURE 2.5 Flows into and out of Benton Cell #3 versus time during September 5–8, 1990. Flows were measured automatically via data loggers; the values were stored as hourly averages. The data points on this graph are six-hour running averages, which smooth out short-term "noise" and emphasize the diel trends. (Data from TVA unpublished data; graph from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

Seepage Losses and Gains

Bank Losses

Shallow seepage, or bank loss, occurs if there is hydrologic communication between the wetland and adjacent aquifers. This is a nearly horizontal flow (see Figure 2.3). If impermeable embankments or liners have been used, bank losses will be negligibly low. However, there are situations where this is not the case, notably for large wetlands treating nontoxic contaminants. An empirical procedure may then be used in which the bank loss is calibrated to the head difference between the water inside and outside of the berm (Guardo, 1999). A linear version of such a model is:

$$Q_b = \lambda L_b (H - H_s) \tag{2.15}$$

where

Q_b = bank seepage flow rate, m^3/d
H = wetland water elevation, m
H_s = external water elevation, m
L_b = length of the berm, m
λ = empirical coefficient, m/d

For instance, wetland levees in southern Florida are typically built from the peat and limestone soils native to the area. Leakage is therefore significant, and has been studied extensively in connection with many canal, storage, and treatment projects. The value used is λ = 15 m/d (Burns and McDonnell, 1992), which represents a very leaky berm.

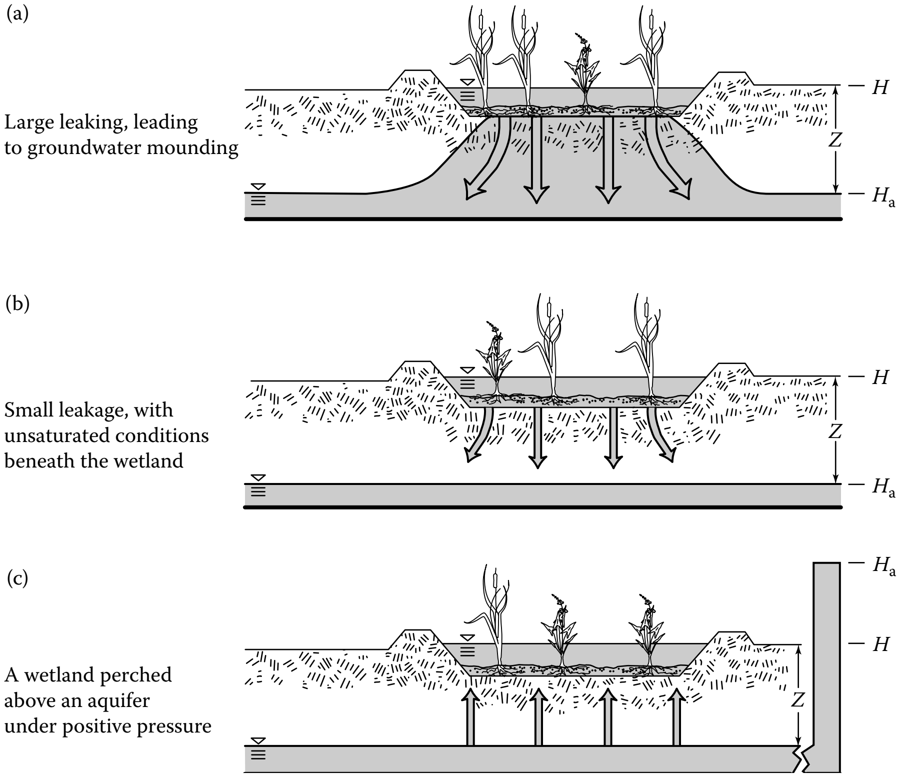


FIGURE 2.6 Three potential groundwater–wetland interactions. (a) Large leakage, leading to groundwater mounding; (b) small leakage, with unsaturated conditions beneath the wetland; (c) a wetland perched above an aquifer under positive pressure. H = stage in the wetland, H_a = piezometric surface in aquifer, and Z = distance from wetland surface to piezometric surface. (Adapted from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

Infiltration

Deep seepage, or infiltration occurs by vertical flow. Unless there is an impermeable barrier, wetland waters may pass downward to the regional piezometric surface (Figure 2.6). The soils under a treatment wetland may range in water condition from fully saturated, forming a water mound on the shallow regional aquifer, to unsaturated flow (trickling).

If the wetland is lined with a relatively impervious layer, it is likely that the underlying strata will be partially dry, with the regional shallow aquifer located some distance below (Figure 2.6b). In this case, it is common practice to estimate leakage from the wetland from:

$$Q_{gw} = kA\left[\frac{H_w - H_{lb}}{H_{lt} - H_{lb}}\right] \tag{2.16}$$

where

A = wetland area, m^2
H_{lb} = elevation of the liner bottom, m
H_{lt} = elevation of the liner top, m
H_w = wetland water surface elevation, m
k = hydraulic conductivity of the liner, m/d
Q_{gw} = infiltration rate, m^3/d

The city of Columbia, Missouri, FWS wetlands provide an example of this situation. It was planned to discharge secondary wastewater to 37 ha of constructed wetlands rather than directly to the Missouri River (Brunner and Kadlec, 1993). Those wetlands were sealed with 30 cm of clay, but were situated on rather permeable soils. The hydraulic conductivity of the clay sealant was 1×10^{-7} cm/s. Water was to be 30 cm deep, and there was 30 cm of topsoil above the clay as a rooting media for wetland plants. Equation 2.17 may be used to estimate a leakage of approximately 0.79 cm/month. Because of the proximity of Columbia's drinking water supply wells, this leakage rate was experimentally confirmed prior to startup. Over a 27-day period, wetland unit one lost 0.21 cm more than the control, indicating a tighter seal than designed.

If there is enough leakage to create a saturated zone under the wetland (Figure 2.6a), then complex three-dimensional flow calculations must be made to ascertain the flow through the wetland bottom to groundwater. These require a substantial quantity of data on the regional water table, regional groundwater flows, and soil hydraulic conductivities by layer. Such calculations are expensive, and usually warranted only when the amount of seepage is vital to the design.

A third possibility is that the wetland is perched on top of, and is isolated from, the shallow regional aquifer. In some instances, such as the Houghton Lake site, the wetland may be located in a clay "dish," which forms an aquiclude for a regional shallow aquifer under pressure (Figure 2.6c). A well drilled through the wetland to the aquifer displays artesian

FIGURE 2.7 Water barrel apparatus to test liner leakage in a VF wetland, Diamond Lake Woods, Minnesota.

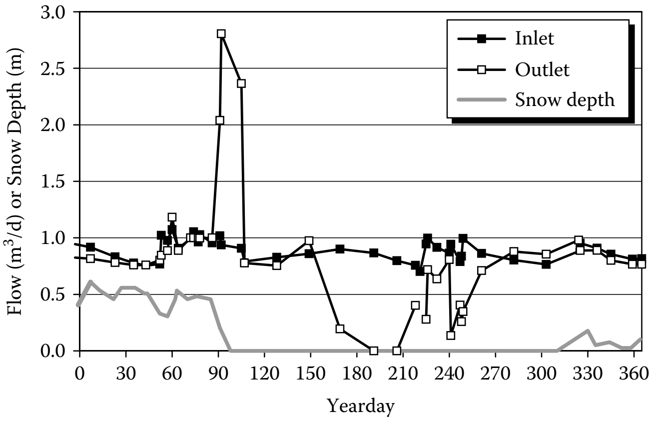


FIGURE 2.9 Flows into and out of NERCC wetland #2 in 1997. The large spike in outflow corresponds to a sudden snowmelt at the end of March. Evapotranspiration losses are apparent in summer. (From Kadlec (2001b) *Water Science and Technology* 44(11/12): 251–258. Reprinted with permission.)

character. The "in-leak" for this system is very small, because the clay layer is many feet thick (Haag, 1979).

In practice, a leak test is often required to demonstrate that a liner in fact performs as designed. One such procedure is known as the Minnesota barrel test (Minnesota Pollution Control Agency, 1989). The water loss from a bottomless barrel placed in the wetland is compared to the water loss from a barrel with a bottom. The barrels collect rain and evaporate water with equal efficiency, so any additional loss from the bottomless barrel must be due to infiltration (Figures 2.7 and 2.8).

Infiltration is allowable in instances where there is not a perceived threat to groundwater quality necessary for the indicated use. That may be drinking water quality, in which case a liner would be used. But the underlying aquifer may have lesser water quality requirements. Such is the case for the Incline Village, Nevada, FWS wetlands, which are underlain by waters with very high concentrations of dissolved evaporites, mostly sulfates. That aquifer is not useable for potable water, and as a consequence, the wetlands were designed to allow infiltration (no liner) (Kadlec *et al.*, 1990). In other situations, the affected groundwater is known to discharge into other water bodies that either provide dilution or further treatment. The former case is typified by the Sacramento wetlands, which leaked about 40% of the added water (Nolte and Associates, 1997). The leakage was known to join a large river, which minimized risks to acceptable levels.

Snowmelt

In northern climates, snowmelt is a springtime component of the liquid water mass balance. The end-of-season snow pack is melted over time, in rough proportion to the temperature excess above freezing. The amount of the snowpack is documented in weather records, such as Climatological Data (NOAA). An example of the effect on flow rate is shown in Figure 2.9, for a HSSF treatment wetland at the NERCC site near Duluth, Minnesota (latitude 46.8°N). The snow depth was about 50 cm in winter, providing insulation enough to prevent freezing of the HSSF wetland bed. A rapid spring

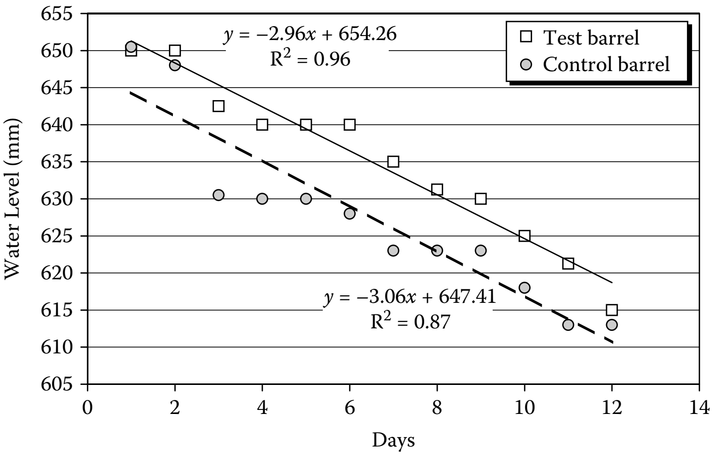


FIGURE 2.8 Results of VF wetland liner testing using the Minnesota barrel method.

thaw created a large spike of melt water that added to the pumped inflow.

Water Storage

The computation of the volume of water stored in a FWS wetland involves the stage-storage curve for the wetland. The derivative of this function is the water surface area:

$$A = \frac{dV}{dh} \tag{2.17}$$

where

A = wetland area, m^2
h = wetland depth, m
V = wetland water volume, m^3

In normal practice, no allowance is made for the volume occupied by vegetation, because of the difficulty of measurement of the vegetation volume. Some wetlands have steeply pitched side slopes, and may be regarded as constant area systems. This implies that the stage-storage curve is a straight line. For instance, Mierau and Trimble (1988) report a nearly linear stage-storage curve for a rectangular diked marsh treating river water. But some wetlands have more complicated topography, such as the treatment wetlands at Des Plaines (Figure 2.10).

This information permits computation of water elevation changes from a knowledge of changes in storage volume. Over any time period, the stage change (ΔH) is given by:

$$\Delta H = \int_{t_1}^{t_2} \frac{dV}{A} = \frac{\Delta V}{A_{avg}} \tag{2.18}$$

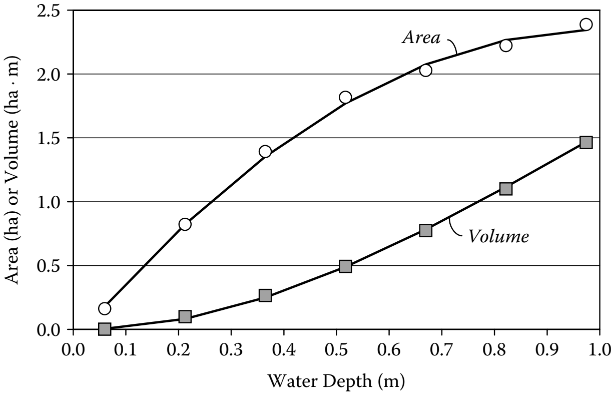


FIGURE 2.10 Stage-storage and stage-area curves for wetland EW3 at Des Plaines, Illinois. The curves are predicted by the following equations. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

$$V = 0.0022 - 0.104h + 2.44h^2 - 0.809h^3$$

$$A = \frac{dV}{dh} = -0.104 + 4.88h - 2.43h^2$$

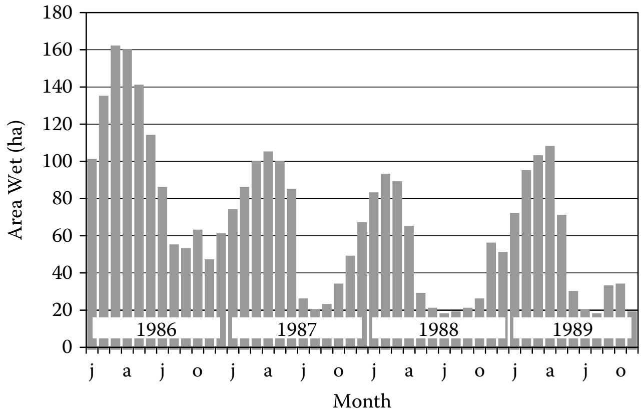


FIGURE 2.11 The expansion and shrinkage of the Incline Village wastewater wetlands as a function of time. Summer water diversions to agricultural uses accelerate the dryout caused by arid conditions. (Data from Kadlec *et al.* (1990) In *Constructed Wetlands in Water Pollution Control*. Cooper and Findlater (Eds.), Pergamon Press, Oxford, United Kingdom, pp. 127–138; graph from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press Boca Raton, Florida.)

where

ΔH = change in wetland water surface elevation, m
ΔV = change in wetland volume, m^3
A_{avg} = mean water surface area over the time period from t_1 to t_2

In the extreme, a wetland may evaporate much of the added water, such as at Incline Village, Nevada. The area of these wetlands responds by expanding and shrinking in response to added water and evapotranspiration (Figure 2.11).

Stormwater treatment wetlands pose a less extreme but important problem: Given fluctuating water levels and wetted areas, what area or volume should be used in pollutant removal calculations? Although this is a complicated question, a bound may be placed on the effective area. If some of the wetland area is dry some of the time, it cannot participate in removals. For a given time period, the number of wetted hectare·days are cumulated, and divided by the total possible wet hectare·days for the entire system footprint to produce the *treatment opportunity fraction*, ϕ (Brown and Caldwell, 1996):

$$\phi = \frac{1}{(t_2 - t_1)A} \int_{t_1}^{t_2} A_{wet}\, dt \tag{2.19}$$

where

A = total wetland area, m^2
A_{wet} = wetland wetted area at time t, m^2
t_1 = start of time period, d
t_2 = end of time period, d
ϕ = treatment opportunity fraction, dimensionless

Event-driven wetlands are discussed in more detail in Chapter 14.

Combined Effects: The Wetland Water Budget

Equation 2.13, the wetland water balance, states that the change in storage in the wetland results from the difference between inflows and outflows. In theory, any one term may be calculated from Equation 2.13 if all the other terms are known. But in practice, none of the measurements are very precise, and large errors may result for such a calculation (Winter, 1981).

Examples of monthly variability of the wetland water budget are given in Table 2.5, for a periphyton pilot wetland (PSTA Test Cell 8) (CH2M Hill, 2001b) and for a large treatment marsh (Boney Marsh) (Mierau and Trimble, 1988). Importantly, the monthly error in closure of the monthly water budget for Boney Marsh ranged from –18% to +7%, with a root mean square (RMS) error of 9% (one outlier removed). These percentages are based upon the combined water inflow. For PSTA Test Cell 8, errors ranged from –30%

TABLE 2.5
Example Water Budgets for FWS Wetlands

Periphyton Test Cell 8
Area: 0.25 ha
Year: 1999
Lined Wetland Cell

Month	Inflow (m^3)	Outflow (m^3)	ET (m^3)	Rain (m^3)	ΔStorage (m^3)	Infiltration (m^3)	Residual (m^3)	Residual (% of Inflow)
January	3,413	4,328	247	75	–797	0	–291	–8%
February	3,378	4,444	272	45	–261	0	–1,031	–30%
March	3,818	4,634	339	267	–118	0	–770	–19%
April	3,803	4,336	340	48	121	0	–946	–25%
May	3,802	3,634	356	14	8	0	–59	–2%
June	3,805	4,421	276	837	16	0	–71	–2%
July	3,807	4,414	358	212	–24	0	–728	–18%
August	3,809	3,615	317	628	81	0	425	10%
September	3,809	5,005	281	453	49	0	–1,074	–25%
October	3,716	4,147	257	932	–57	0	301	6%
November	3,889	4,418	222	29	–63	0	–659	–17%
December	3,841	3,065	185	100	36	0	635	16%
Average	**3,741**	**4,205**	**287**	**303**	**–84**	**0**	**–356**	**–9%**
RMS Residual								**17.3%**

Boney Marsh
Area: 49 ha
Year: 1983
Unlined Wetland Cell

	Inflow (1,000 m^3)	Outflow (1,000 m^3)	ET (1,000 m^3)	Rain (1,000 m^3)	ΔStorage (1,000 m^3)	Seepage (1,000 m^3)	Residual (1,000 m^3)	% Error
January	335	395	30	27	–37	3	–28	–7.8%
February	313	362	37	92	6	3	–2	–0.5%
March	340	418	55	59	–4	3	–73	–18.2%
April	322	392	62	22	–65	3	–48	–13.8%
May	66	88	84	10	–27	3	–72	–95.1%
June	239	199	61	136	110	3	2	0.6%
July	321	281	67	43	–12	3	25	6.9%
August	354	277	50	45	74	3	–6	–1.4%
September	384	259	43	47	108	3	18	4.2%
October	356	411	43	18	–18	3	–66	–17.6%
November	303	403	33	16	–115	3	–3	–1.0%
December	374	399	27	43	–1	3	–11	–2.7%
Average	**309**	**324**	**49**	**47**	**1**	**3**	**–22**	**–7.1%**
RMS Residual								**9.0%**

to +16%, with a root mean square error of 17%. The RMS error increases with decreasing water budget period. For Boney Marsh, over an eight-year period, the daily, monthly, and annual RMS errors were 67%, 16%, and 7%, respectively (Mierau and Trimble, 1988).

These are not extreme examples. Similar lack of closure has been reported for four wetlands at Sacramento, where all mass balance terms were measured independently, including infiltration measured by drawdown (Nolte and Associates, 1998b). The RMS monthly errors were 60%, 47%, 26%, and 19% for Cells 3, 5, 7, and 9, respectively. The annual percentage residuals were −56%, −44%, −15%, and −15%, respectively. The conclusion was that these apparent water losses were due to faulty inflow or outflow measurements.

These examples serve to alert the wetland designer or operator that care must be taken in water flow measurements and that water balance differencing is apt to provide estimates with large uncertainty. With great care, balance closure may be held to the ±5 to 10% range (Mierau and Trimble, 1988; Guardo, 1999; Martinez and Wise, 2001).

2.2 FWS WETLAND HYDRAULICS

Early in the history of research and development related to overland flow in wetlands, mathematical descriptions were often adaptations of turbulent open channel flow formulae. These are discussed in detail in a number of texts—for example, the work of French (1985). The general approach is utilization of mass, energy, and momentum conservation equations, coupled with an equation for frictional resistance. Perhaps the most common friction equation is Manning's equation, which will be further discussed later in this section.

There is a fundamental problem with the utilization of Manning's equation to wetland surface water flows: Manning's equation is a correlation for turbulent flows, whereas FWS wetlands are nearly always in a laminar or transitional flow regime (based on open channel flow criteria). Under these conditions, Manning's n is not constant, but is strongly velocity dependent (Hosokawa and Horie, 1992). There is also a difficulty with the extension of open channel flow concepts to densely vegetated channels. The frictional effects that retard flow in open channels are associated primarily with drag exerted by the channel bottom and sides. Wetland friction in dense macrophyte stands is dominated by drag exerted by the stems and litter, with bottom drag playing a very minor role.

As a consequence, overland flow parameters determined from open channel theory are not applicable to wetlands. In particular, Manning's coefficient is no longer a constant; it depends upon velocity and depth as well as stem density. Predictions from previous information on nonwetland vegetated channels are seriously in error (Hall and Freeman, 1994). Unfortunately, much of the existing information on wetland surface flow has been interpreted and reported via Manning's equation, and so it cannot be avoided.

Major advances in formulating correct and improved approaches to overland flow in wetlands have been made in the past ten years (e.g., Nepf, 1999; Oldham and Sturman, 2001; Choi *et al.*, 2003). This section utilizes the emerging knowledge and calibration database to provide methods to predict depths and velocities in FWS wetlands.

THE CALCULATION STRUCTURE

Wetland water depths and flow rates are controlled by two major wetland features; the outlet structure and resistance to flow within the wetland. In general, it is very desirable to have control at the outlet structure, because then the operator has control over water depth. Under complete outlet control, a level pool of water exists upstream of the outlet structure, regardless of what is growing there. However, that is not always possible, particularly for large or densely vegetated wetlands. Water may be held up by the vegetation at a depth that is independent of the outlet structure setting.

Four different situations may occur, and are easily visualized (Figure 2.12):

1. Very low flow; complete weir control. There is a level pool upstream of the outlet structure, and wetland water stage is spatially invariant.
2. Partial weir control (*M1 profile*). There is a level pool in the region near the weir, but a gradient in stage near the wetland inlet. This is a distance thickening sheet flow.
3. Normal depth flow. Vegetation drag controls the depth to exactly the stage created by the weir.
4. Large flow; partial weir control (*M2 profile*). There is a constant depth flow, at the normal depth, near the inflow, followed by decreasing depth near the outflow. This is a distance thinning sheet flow.

These various possibilities are covered by a backwater calculation. Because wetlands nearly always meet the criterion for gradually varied flow with a small Froude number (French, 1985), the water flow momentum balance can be

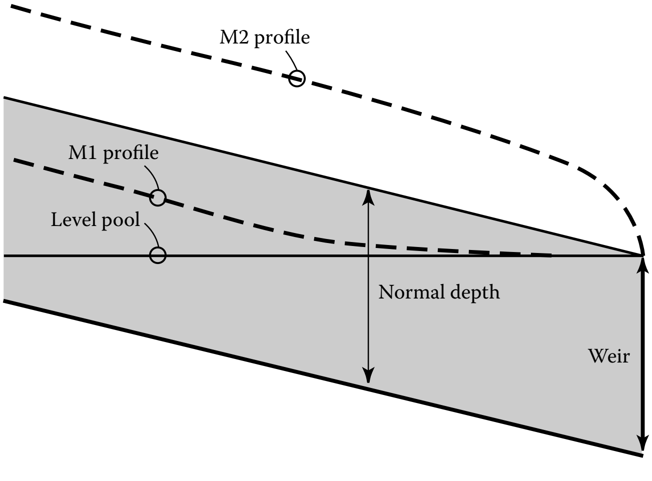


FIGURE 2.12 FWS water surface profiles for a fixed height over the outlet weir and various inlet flows. The notation follows French (1985).

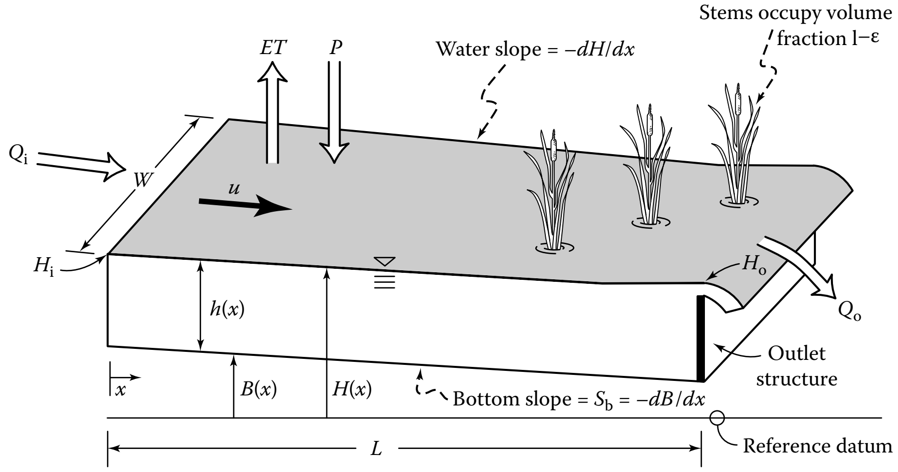


FIGURE 2.13 Notation for FWS bed hydraulics. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

simplified to contain only gravitational and friction terms. The component pieces are the spatial water mass balance, the friction equation, and specification of inflow, geometry, and outlet depth setting. For one-dimensional (rectangular) systems, *in the absence of rain or ET effects*, the flow situation can be simplified as indicated in Equation 2.20. Notation is given in Figure 2.13.

The spatial water mass balance, water depth (h), and superficial flow velocity (u) are distance-variable:

$$\frac{dQ}{dx} = \frac{d(hWu)}{dx} = 0 \tag{2.20}$$

where
- h = water depth, m
- Q = volumetric flow rate, m^3/d
- W = wetland width, m
- u = superficial flow velocity, m/d
- x = distance in flow direction, m

Frictional losses can be represented by a general power law relationship. This is discussed in further detail in the next section of this chapter:

$$u = a\ h^{(b-1)} S^c \tag{2.21}$$

where
- a, b, c = friction parameters
- u = superficial water velocity, m/s
- h = water depth, m
- $S = -dH/dx$ = negative of the water surface slope, m/m

Water elevation is water depth plus bed bottom elevation profile (Figure 2.13):

$$H = B + h \tag{2.22}$$

where
- B = elevation of the bed bottom above datum, m
- H = elevation of the water surface, m
- h = water depth, m

Equations 2.20, 2.21, and 2.22 combine to give:

$$aWh^b\left(-\frac{d(h+B)}{dx}\right)^c = Q \tag{2.23}$$

The boundary condition necessary to solve Equation 2.23 is typically a specification of the outlet water level, as determined by a weir or receiving pool:

$$H_{at\ x=L} = H_o \tag{2.24}$$

where
- H = elevation of the water surface, m
- x = distance in flow direction, m
- L = length of wetland cell along flow path, m

Equations 2.23 and 2.24 cannot be solved analytically to a closed-form answer, but numerical solution is easy via any one of a number of methods. Required input parameters are the bottom slope profile, the flow rate and the height over the weir, together with the friction parameters a, b, c.

Although there can be any of several types of outflow structure, it is useful to illustrate the determination of the weir overflow stage for that choice of outlet control. The commonly used equation for a rectangular weir is:

$$Q_o = C_E\, W_W\, (H_o - H_W)^{1.5} \tag{2.25}$$

where
- Q_o = outlet flow rate, m^3/d
- C_E = weir discharge coefficient, (m^3/d)/(m$^{2.5}$)
- W_W = width of weir, m
- H_o = water surface elevation at wetland outlet, m
- H_W = weir crest elevation, m

Friction Equations for FWS Wetland Flows

All of the required information for the backwater calculation is readily obtainable, except for the friction parameters *a*, *b*, and *c*. Water flow through the wetland is associated with a local frictional head loss, given by Equation 2.21. This is a power law representation of the fact that the water velocity is related to the water surface slope ($S = -dH/dx$) and to the depth of the water (h). This generalized form of Equation 2.21 was first suggested by Horton (1938). He proposed *b* equal to zero for vegetated flow, 2.0 for laminar flow, and 4/3 for turbulent flow; and *c* equal to 1.0 for laminar or vegetated flow and 2.0 for turbulent flow; and $a = 1/K$ is a constant (different for the three cases). Transition flows were to be handled by adjusting the value of *b* between 1.0 and 2.0. We use this form here, although for reasons different from Horton (1938), as will be explained below.

The friction equation is a vertically averaged result, based upon a reluctance to go to the complexity of three-dimensional computational fluid mechanics. This results in two difficulties in the wetland environment:

1. There is a vertical profile of vegetation resistance in many cases, because the submerged plant parts are often stratified.
2. A good deal of the literature presumes flows in evenly flat-bottomed systems, which is not the case for wetlands. It is usual to have a significant amount of microtopographical relief in the wetland, which also factors into vertical averaging.

Flows Controlled by Bottom Friction

The framework that is very often borrowed from the literature is adaptation of constant depth, open channel flow equations. It is to be noted that *this situation should not apply to vegetated wetlands*, but that has not prevented widespread use of the equations.

When $a = 1/K$, $b = 3$, and $c = 1$, Equation 2.21 becomes the equation for laminar flow in an open channel as shown in Equation 2.26 (Straub *et al.*, 1958):

$$u = \frac{1}{K} h^2 S \tag{2.26}$$

where
K = laminar flow friction coefficient, s·m
u = superficial flow velocity, m/s
h = water depth, m
$S = -dH/dx$ = negative of the water surface slope, m/m

Note that a unit conversion is necessary to convert to the mass balance unit of days. The limit of this formulation for a channel devoid of vegetation is the *depth Reynold's number* (Re) less than 2,500:

$$\text{Re} = \frac{h\rho u}{\mu} < 2{,}500 \tag{2.27}$$

where
Re = depth Reynold's number, dimensionless
h = water depth, m
u = superficial water velocity, m/s
μ = viscosity of water, kg/m·s
ρ = density of water, kg/m^3

For average warm water properties and a typical water depth of 30 cm, a Reynold's number of less than 2,500 translates to flow velocities less than about 700 m/d, a range that includes most FWS wetlands, except for the very largest.

When $a = 1/n$, $b = 5/3$, and $c = 1/2$, Equation 2.21 becomes Manning's equation (French, 1985):

$$u = \frac{1}{n} h^{2/3} S^{1/2} \tag{2.28}$$

where
n = Manning's coefficient, s/m$^{1/3}$
u = superficial water velocity, m/s
h = water depth, m
$S = -dH/dx$ = negative of the water surface slope, m/m

Note that a unit conversion is again necessary to convert to the mass balance unit of days.

Suppose that open channel information were to be used to estimate Manning's n for a wetland. Guidance may be found in estimation procedures in the hydraulics literature, for instance French (1985). The value of n may be estimated from information on the channel character, type of vegetation, changes in cross section, surface irregularity, obstructions, and channel alignment. Using the highest value of every contributing factor, the maximum open channel n value is 0.29 s/m$^{1/3}$ (French, 1985). This is approximately one order of magnitude less than values determined from actual wetland data. *Clearly, open channel, turbulent flow information is inadequate to describe the densely vegetated, low-flow wetland environment.*

Nepf (1999) used both laboratory flumes and field measurements in a Spartina marsh to conclude that bed drag is negligible compared to stem and leaf drag at densities of submerged vegetation of one percent by volume and higher. *Therefore, Equations 2.26 and 2.28 are both inappropriate for vegetated wetlands.*

Flows Controlled by Stem Drag

The presence of submerged stems, leaves, and litter creates an underwater environment dominated by drag on those surfaces, rather than the channel bottom. The common measures of vegetation density are the number per square meter times their diameter:

$$a = n_s d \tag{2.29}$$

$$ad = n_s d^2 \tag{2.30}$$

where

a = projected plant area normal to flow per unit volume, m^2/m^3
d = cylinder diameter of vegetation, m
ad = fraction of volume occupied by plants, m^3/m^3
n_s = number of stems per unit area, $\#/m^2$

The traditional measure of vegetative surface area is the leaf area index (LAI). In the context of immersed surfaces and drag, it is the fraction of the total LAI below water and its vertical distribution that are of interest. Although LAI and area normal to flow are not identical, a direct relation between them would be expected.

The resistance to flow through this submersed matrix is described by a drag equation (Nepf and Koch, 1999):

$$S = C_D\, a\left(\frac{u^2}{2g}\right) \quad (2.31)$$

where

C_D = drag coefficient, dimensionless
$S = -dH/dx$ = negative of the water surface slope, m/m
a = projected plant area normal to flow per unit volume, m^2/m^3
u = superficial water velocity, m/s
g = acceleration of gravity, m/s^2

If the *stem Reynolds number* (Re_s) within the array is less than 200, the flow will be laminar:

$$Re_s = \frac{d\rho u}{\mu} < 200 \quad (2.32)$$

where

Re_s = stem Reynold's number, dimensionless
d = cylinder diameter of vegetation, m
u = superficial flow velocity, m/s
μ = water viscosity, kg/m·s
ρ = water density, kg/m^3

As a point of reference, stems of one cm diameter in a flow of 1,000 m/d would produce an $Re_s = 116$, which is still within the laminar flow range. For flow velocities typically encountered in FWS wetlands, this implies that flows proceed with interfering laminar wakes (Nepf, 1999). Stem densities are such that drag is determined by obstruction of flow (form drag). For this circumstance,

$$C_D = \frac{K_1}{Re_s} \quad (2.33)$$

where

C_D = drag coefficient, dimensionless
K_1 = constant, unitless
Re_s = stem Reynold's number, dimensionless

Under these circumstances, it may be shown that yet another set of parameters might be applicable in Equation 2.21, i.e., $b = 1$ and $c = 1$:

$$u = \frac{K_{stem}}{n_s} S \quad (2.34)$$

where

u = superficial flow velocity, m/s
n_s = number of stems per unit area, $\#/m^2$
K_{stem} = conveyance coefficient, $m^{-1}\cdot s^{-1}$
$S = -dH/dx$ = negative of the water surface slope, m/m

Note that a unit conversion is again necessary to convert to the mass balance unit of days. There is no depth effect in this formulation, which is, in effect, Darcy's law for uniform porous media, where the porous media in this case is a bed of submerged vegetation. Data from channels with vertical rods indeed support this analysis (Nepf, 1999; Schmid *et al.*, 2004b). Hall and Freeman (1994) confirmed the direct proportionality of resistance to stem density for bulrushes, which have a plant geometry very similar to vertical rods.

There are, however, several other important features of wetland flows that must be taken into account. There are vertical and spatial profiles of stem-leaf density, wind forces can move water (Jenter and Duff, 1999), and the wetland bottom is not flat (Kadlec, 1990).

Vertical Profiles of Stem Density

The vertical location of plant stems and leaves varies with the type of vegetation under consideration. One limiting case is floating plants, such as water hyacinths (*Eichhornia crassipes*), which populate only the topmost stratum of the water column. Rooted plants with floating leaves, such as water lilies (*Nymphaea* spp.), also place most drag in the vicinity of the water surface, with a lesser amount in the water column due to stems. In contrast, most of the commonly used emergent macrophytes in treatment wetlands have stems and/or leaves distributed throughout the water column, but the distributions are not necessarily uniform. A bottom layer normally contains dead and prostrate plant parts, which is the litter layer. Stems or culms are dominant portion of these lower horizons. Bulrushes continue with stem morphology exclusively, but leaves are dominant at mid-depths for cattails, sedges, and reeds. In combination, the distributions of drag surfaces, for many emergent marsh systems, are fairly uniform over typical operating depth ranges (Figure 2.14), as indicated by the linearity of the cumulative LAI with depth. Thus, in the absence of any other factors, flow would be expected to follow a stem/leaf drag relationship such as Equation 2.34.

The Influence of Bathymetric Variability

The bottom elevation of many FWS wetlands is irregular, with local depressions and hummocks. On a large scale, these are

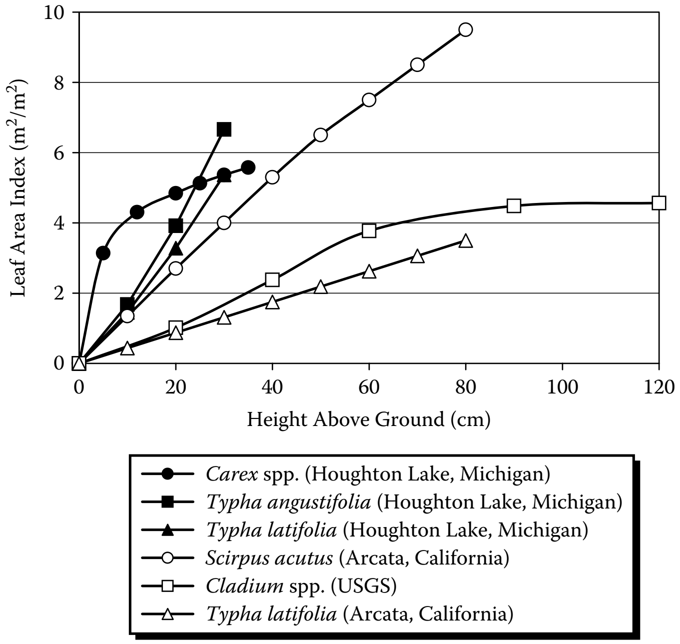


FIGURE 2.14 Leaf area indices for various emergent macrophytes. These are cumulative numbers, representing the total leaf area below a given elevation above ground. (Data for USGS: Rybicki *et al.* (2000) *Sawgrass density, biomass, and leaf area index a flume study in support of research on wind sheltering effects in the Florida Everglades*. Open File Report 00-172, U.S. Geological Survey: Reston, Virginia; for Arcata: U.S. EPA (1999) *Free water surface wetlands for wastewater treatment: A technology assessment*. EPA 832/R-99/002, U.S. EPA Office of Water: Washington, D.C. 165 pp.; for Houghton Lake: unpublished data; and Kadlec (1990) *Journal of the Hydraulics Division (ASCE)* 116(5): 691–706.) Corresponding porosities were:

Carex spp.	Houghton Lake	99%
Typha angustifolia	Houghton Lake	96%
Typha latifolia	Houghton Lake	96%
Scirpus acutus	Arcata	97%
Cladium spp.	USGS	98%
Typha latifolia	Arcata	93%

quantified by depth–area–volume relations (see Figure 2.10). On a small scale, these features define the micro-topography of the wetland bottom, and are represented by a soil surface elevation distribution. Small constructed wetlands are typically designed to be graded at a specified tolerance, such as ±5 cm. In practice, these tolerances often either are not achieved during construction, or change as the bottom of the wetland accumulates sediments and plant detritus over time (Figure 2.15). Interestingly, some natural wetlands have about the same fine-scale distributions of soil elevations as do constructed wetlands.

The effect of such uneven bottoms upon the friction model depends upon the orientation and shape of the high spots and depressions (Stothoff and Mitchell-Bruker, 2003). Ridge features may either be parallel to flow, and act as flow-straighteners, or be perpendicular to flow and act as "speed bumps." To illustrate the potential effects, assume the bottom elevation distribution represents the flow cross section (Kadlec, 1990; Choi *et al.*, 2003). In order that water depth remain positive, depth is measured with respect to the lowest soil elevation. A purely geometric effect prevails: there is not much cross section available for flows at very low

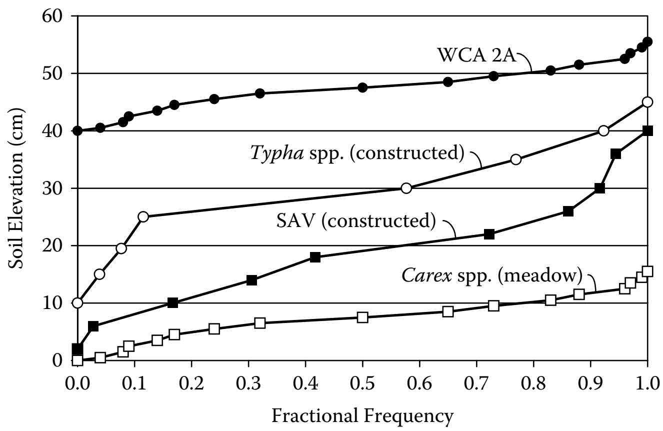


FIGURE 2.15 Bottom elevation variability in natural and constructed wetlands, measured on a 10–20 m spacing. The datum in each case is arbitrary, and has been adjusted to provide vertical separation of the curves.

TABLE 2.6
Speed and Direction of Water Movement at Various Depths in a Sparsely Vegetated Marsh, WCA2A

Depth (cm)	Speed (cm/s)	Direction (degrees)
5	0.42	103
19	0.25	44
25	0.30	11
34	0.40	170
39	0.31	246
44	0.25	108

Source: Data from Romanowicz and Richardson (1997) *Hydrologic investigation of water conservation area 2-A*, Chapter 12 in the 1996-1997 Biennial Report to the Everglades Agricultural Area Environmental Protection District, Publication 97-05, Duke Wetland Center: Durham, North Carolina.

water stage. This effect is not lost until water depths are well above the point of complete inundation. Application of Equation 2.34 to a linear distribution produces a depth effect on conveyance capacity. For example, a straight line approximation to the cattail data in Figure 2.14, applied to flows at depths up to 60 cm, introduces depth dependence as represented by $b = 1.94$ in the general Equation 2.21, with $c = 1.0$.

Wind Effects

Densely vegetated emergent FWS wetlands provide shelter from wind and minimize wind-driven water flow. However, the same is not true for open water areas, with or without submergent vegetation. It is possible to assess the potential for wind driven flow by comparing the drag force created by wind to that created by drag on submerged plant parts. For instance, at a wind speed of 5 m/s, for 200 one-cm stems per square meter and a water depth of 30 cm, wind drag is three times as strong as stem drag (based on Teeter *et al.*, 2001). As a consequence, surface water moves in the direction of the wind, with compensatory flows in lower water regions (Table 2.6). As yet, there is no practical predictive method of dealing with wind friction, and it therefore contributes to the variability of marsh friction calibrations.

Wetland Data

Generalized Friction Parameters

It would be desirable to have predictive methods for the parameters a, b, and c in friction equations such as Equation 2.21. At the present time, data exist for only a few wetlands (Table 2.7). As discussed above, site-specific factors are known to be very important, and it is very dangerous to extrapolate from nonwetland information. Manning's coefficient is clearly not constant for the wetland environment, and it is preferable to utilize a model which describes the depth variability, namely Equation 2.21. The exponent c is 0.5 in the turbulent open channel formulation. However, investigations on wetland systems indicate a higher value of c is appropriate. As a limiting value, laminar flow around a uniform array of submerged objects over a flat bottom is theoretically described by $c = 1.0$. Until more data becomes available, a value of $c = 1.0$ is recommended.

The exponent b is 1.67 in the turbulent open channel formulation. But the depth variability measured for wetlands increases this value, due to bottom irregularity and other factors. Until more data becomes available, a value of $b = 3.0$ is recommended for FWS wetland treatment systems. The coefficient a remains a function of vegetation and litter density. Until more data become available, a value of $a = 1.0 \times 10^7$ $m^{-1}d^{-1}$ is recommended for densely vegetated wetlands, and $a = 5.0 \times 10^7$ $m^{-1}d^{-1}$ is recommended for sparsely vegetated wetlands.

Summary of recommended of recommended parameters for the generalized FWS friction relationship (Equation 2.21):

$$a = 1.0 \times 10^7 \ m^{-1}d^{-1} \text{ (densely vegetated)}$$

$$a = 5.0 \times 10^7 \ m^{-1}d^{-1} \text{ (sparsely vegetated)}$$

$$b = 3.0$$

$$c = 1.0$$

TABLE 2.7
Friction Equation Coefficients for FWS Wetlands

Vegetation	Depth Range (cm)	Depth Exponent, *b*	Slope Exponent, *c*	Conveyance Coefficient, *a* (m/d)/m$^{(b-1)}$	Reference
Sedge	0.05–0.25	3.00	0.71	2.00E + 08	Kadlec *et al.* (1981)
Sedge	0.08–0.30	2.50	1.00	5.00E + 07	Kadlec (1990)
Sparse emergents	0.20–0.80	1.44	1.00	6.20E + 06	Bolster and Saiers (2002)
Sparse cattails	0.30–0.85	1.60	1.00	1.80E + 07	Choi *et al.* (2003)
Sparse sawgrass	0.30–0.85	1.64	1.00	4.70E + 07	Choi *et al.* (2003)
Cattail	0.05–0.21	3.00	1.00	6.00E + 07	Hammer and Kadlec (1986)
Cattail	0.05–0.21	2.00	1.00	9.00E + 06	Hammer and Kadlec (1986)

TABLE 2.8
Values of Manning's *n* Measured for FWS Wetlands

Project	Vegetation	Depth (m)	Velocity (m/d)	Reynolds Number	Manning's *n* (s/m$^{(1/3)}$)	Source
ENR Cell 1	Cattails + SAV	0.36–0.79	30–867	125–7,900	0.43–2.50	Unpublished data SFWMD
ENR Cell 4	SAV	0.36–0.81	277–1,562	351–4,265	0.42–1.33	Unpublished data SFWMD
Sacramento Cell 3	Dense bulrush	0.45–0.60	50–60	257–448	5.9–6.7	Dombeck *et al.* (1998)
Sacramento Cell 7	Dense bulrush	0.45–0.60	40–75	367–928	2.1–7.6	Dombeck *et al.* (1998)
Benton Cell 1	Cattail	0.17–0.35	400	770–1,070	13.8	Unpublished data TVA
Benton Cell 2	Woolgrass	0.12–0.42	110–358	520	3.3	Unpublished data TVA
Lewisville, Texas, Flume	Dense bulrush	0.10–0.43	2,075–13,400	7,000–47,500	0.16–0.93	Freeman *et al.* (1994)
Stennis Space Flume	Sawgrass	0.15–0.75	132–3,950	460–23,000	0.32–1.80	Jenter and Schaffranek (2001)
Boney Marsh	Mix	0.30–0.70	35–135	108–713	1–4	Mierau and Trimble (1988)
Chandler Slough	Water hyacinth	0.40–0.70	—	—	0.20–0.55	Shih and Rahi (1982)
Chandler Slough	Pickerel weed, buttonbush	0.35–0.65	—	—	0.18–0.47	Shih and Rahi (1982)
Shark River Slough	Sparse emergents	0.10–0.60	—	—	0.40–2.50	Rosendahl (1981)
WCA1	Sawgrass	0.15–1.50	—	—	0.33–1.20	Shih *et al.* (1979)
WCA2A	Sawgrass	0.15–1.50	—	—	0.32–1.20	Shih *et al.* (1979)
Chandler Slough	Mix	0.15–1.50	—	—	0.29–0.68	Shih *et al.* (1979)

Manning's Coefficients

Although not appropriate for FWS wetlands, Manning's Equation (2.28) has, nevertheless, been widely used and calibrated in FWS wetlands (Table 2.8). Florida emergent marsh studies comprise a large fraction of the available wetland friction information. These serve to provide general guidelines for site-specific factors.

Generally, Manning's *n* is strongly depth dependent for FWS systems, decreasing as depth increases. The nature of this dependence is illustrated in Figure 2.16 for two Florida marsh studies. Over a depth range of 30–90 cm, Manning's *n* decreased by a factor of five for an emergent and submerged aquatic vegetation (SAV) wetland, and by a factor of three for a SAV-only wetland. This is somewhat surprising, because open channel theory predicts an increase in *n* with increasing depth. Although that theoretical result has not been observed in treatment wetlands, there are examples of lesser depth dependence, such as the Boney Marsh FWS wetlands. Mierau and Trimble (1988) found no depth dependence of *n* in an eight-year data analysis. Shih *et al.* (1979) found only a factor of two decrease over a depth range of 30–90 cm.

Likewise, *n* values are dependent on vegetation density, because stems and litter provide the dominant drag surfaces. A linear relationship was found for *Schoenoplectus* (*Scirpus) validus* (Hall and Freeman, 1994). Therefore it is not surprising to find a strong seasonal dependence of *n*, because vegetation

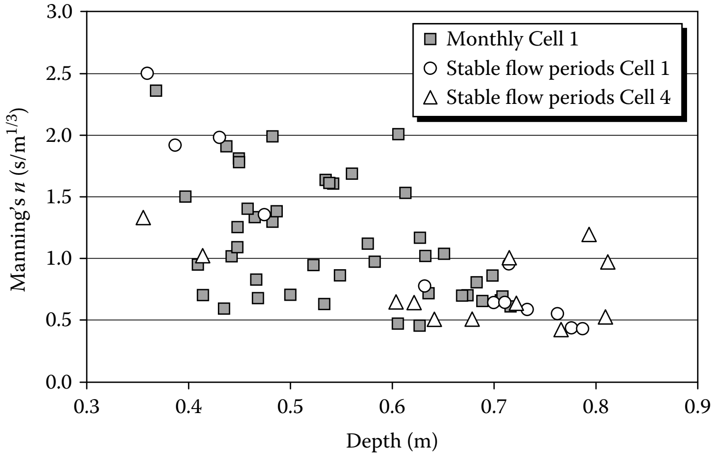


FIGURE 2.16 Manning's *n* versus depth for ENR project Cells. Cell 1 was an emergent-SAV mix; Cell 4 was SAV. Data span four years. Stable flow periods are at constant flow, monthly values include changes in storage and flow.

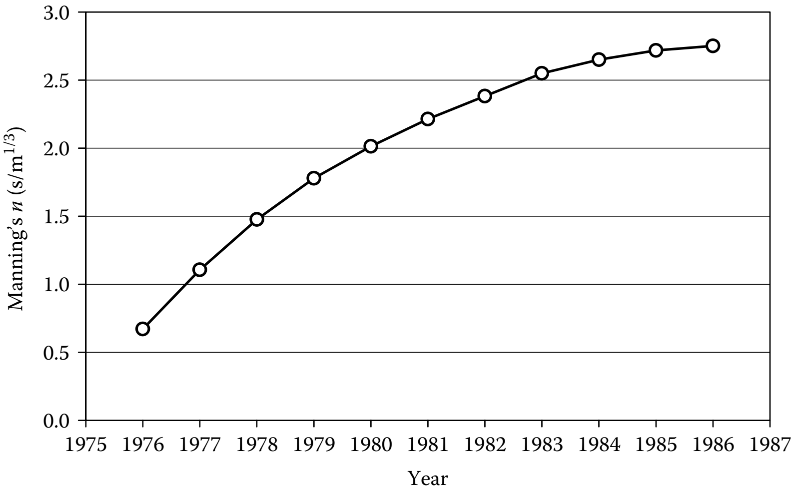


FIGURE 2.17 The increase of friction in a developing wetland, as shown by the central tendency of each year's data. Vegetation densities in Boney Marsh may have been only part of the reason for increases: increases in litter and sediment accumulation could have altered these measurements. (Data from Mierau and Trimble (1988) *Hydrologic characteristics of the Kissimmee River Floodplain Boney Marsh Experimental Area*. Technical memorandum, South Florida Water Management District (September 1988); graph from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

changes seasonally (Shih and Rahi, 1982). Because both litter and live stems are involved, the relation is not easily predictable; it depends on litterfall events.

The progress of a constructed system from an initial sparse vegetation to a more densely vegetated condition is accompanied by increases in the friction coefficient. Boney Marsh, Florida, received pumped river water over several years beginning in 1976. Hydrologic studies produced weekly values of Manning's n (Mierau and Trimble, 1988). The biological dynamics of the Boney marsh operation produced considerable scatter in Manning's n, but the year-to-year trend line was upward from 0.6 to 2.7 s/m$^{1/3}$ (Figure 2.17).

Head Loss Calculations

The implementation of Equation 2.23 requires numerical integration, which is inconvenient in conceptual design calculations. But because of the extreme nonlinearity of the equations, it is very inaccurate to use average values. Accordingly, it is better to use precalculated values of the head loss for the intended design conditions. To accomplish this, the case of a rectangular constructed wetland is considered, with a negligible loss or gain of water due to P and ET. Equation 2.23 is de-dimensionalized using the wetland length and the outlet water depth:

$$y = \frac{h}{h_o} \qquad y_b = \frac{B}{h_o} \qquad z = \frac{x}{L} \qquad S_1 = \frac{dy_b}{dz} \tag{2.35}$$

$$y^3 = \left(\frac{d(y+y_b)}{dz}\right) = y^3\left(-\frac{dy}{dz} - S_1\right) = \frac{qL^2}{ah_o^4} = M_1 \tag{2.36}$$

where

h = water depth, m
h_o = water depth at outlet, m
B = elevation of the bed bottom above datum, m
L = wetland length, m
q = hydraulic loading rate, m/d

and the rest of the new variables are defined in Equations 2.35 and 2.36. It is presumed that the outlet water depth is fixed. Integration of Equation 2.36 yields the inlet water depth, and hence the head loss for a given wetland. Solutions depend on two parameters: S_1, which represents the bed slope, and M_1, which contains the friction coefficient, the hydraulic loading rate, outlet depth, and wetland length. Figure 2.18 presents the solution of Equation 2.36 for different parameter values. It may be used to estimate head losses in FWS wetlands.

An Example

A surface flow wetland is to be built to treat 200 m^3/d of secondary municipal wastewater. The appropriate hydraulic loading rate has been determined to be 2 cm/d. Site considerations indicate that a length of 400 meters is desirable. A bed slope of 20 cm over the 400-m length is to be used to provide drainage. The outlet weir is to be set to maintain 20 cm depth at the outlet. What is the estimated head loss?

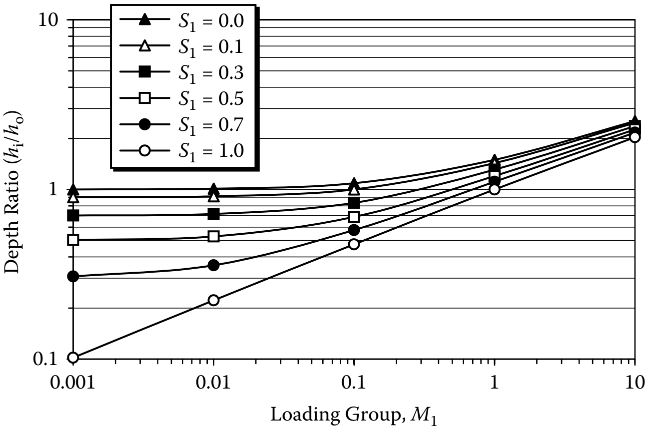


FIGURE 2.18 Inlet/outlet depth ratio for FWS wetlands of different slopes and different loading rates. The friction power law is used, with $b = 3$ and $c = 1$. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

$$S_1 = \frac{L}{h_0}\left(-\frac{dB}{dx}\right) \quad M_1 = \frac{qL^2}{h_0^4 a}$$

The constants needed to use Figure 2.18 are:

$$a = 1 \times 10^7 \ \text{m}^{-1}\text{d}^{-1}$$
$$dB/dx = 0.20 / 400 = 0.0005$$
$$h_o = 0.20 \text{ m}$$
$$S_1 = 1.0$$
$$q = 0.02 \text{ m/d}$$
$$L = 200 \text{ m}$$
$$M_1 = \frac{(0.02)(400)^2}{(0.2)^4(1 \times 10^7)} = 0.2$$

Referring to Figure 2.18, the ratio of inlet depth to outlet depth is 0.6. Therefore:

$$h_i = (0.6)(0.20) = 0.12 \text{ m}$$
$$H_i = B_i + h_i = 0.20 + 0.12 = 0.32 \text{ m}$$
$$\Delta H = 0.32 - 0.20 = 0.12 \text{ m} = 12 \text{ cm}$$

2.3 HSSF WETLAND HYDRAULICS

The idea of flowing water through a planted bed of porous media seems simple enough; yet numerous difficulties have arisen in practice. Sometimes these problems have been traced to incorrect design calculations; at other times problems have resulted from changes in the conditions in the bed. A great deal of confusion has been evidenced regarding the movement of water through HSSF wetlands. Rules of thumb abound in the literature, many of which do not acknowledge the simple physics of water movement. The literature is replete with misapplications of the fundamental relations between head loss and flow rate. In this section, relevant calculations are examined and bounds placed on the variables governing the ability of wetlands to operate in subsurface flow with rooted macrophytes.

Prior to 1995, gravel bed HSSF wetlands in the United States were frequently observed to be flooded (Kadlec and Knight, 1996). The two leading causes were clogging of the media and improper hydraulic design. The same appeared to be true for other countries as well (Brix, 1994a), especially HSSF wetlands that used soil for the bed medium. Flooded HSSF systems have been tolerated in many instances because the hydraulically failed mode of flooded operation is the FWS wetland, which may provide treatment performance nearly as efficient as the HSSF wetland.

FLOW IN POROUS MEDIA

There is a very long history of research and development related to flow in porous media. Descriptions of flow phenomena started with the propositions of Darcy in 1856 (Brown, 2002), and have grown to include several texts on the subject. Several types of flow can occur in general; here the concern is solely for the case of fully saturated flow with an unconfined top interface with air, either in or above the bed. Full saturation refers to the absence of a capillary fringe, in which both air and water occupy the voids between particles.

HSSF wetlands operate in thin sheet flow, with a free upper surface. Flows may be averaged over the vertical (thin) dimension, for the case of the upper surface exposed to the atmosphere, to yield the one-dimensional Dupuit–Forcheimer equation:

$$\frac{\partial(\varepsilon H)}{\partial t} = \frac{\partial}{\partial x}\left(kH\frac{\partial H}{\partial x}\right) + P - ET \tag{2.37}$$

where
ET = evapotranspiration loss, m/d
P = precipitation, m/d
k = hydraulic conductivity, m/d
H = elevation of the free water surface, m
x = longitudinal distance, m
ε = porosity, dimensionless

It is important to note that this equation embodies the assumption that the driving force for flow is a tilt to the water surface ($\partial H/\partial x$). A simpler version of this theory will suffice for HSSF wetland design purposes.

ADAPTATIONS FOR HSSF WETLANDS

The following developments presume that the wetland is in a steady state condition, but later it will be shown that this is rarely the case. The representation will therefore be for long-term, average performance. It is presumed that the porous medium is isotropic. This is probably not true, due to the presence of plant roots and other introduced particulates. The variability in the vertical and transverse directions is accounted for by averaging. Longitudinal variations in

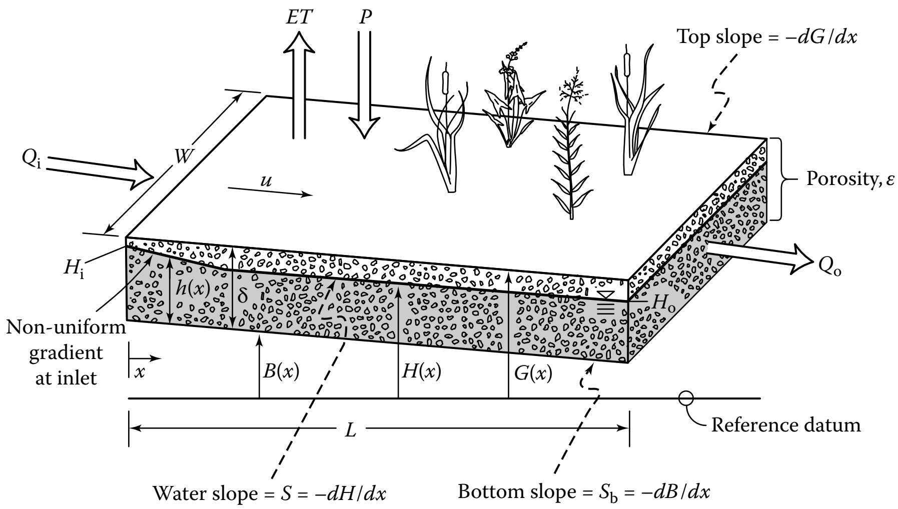


FIGURE 2.19 Notation for HSSF bed hydraulic calculation for the simplest case. The actual velocity of water is $v = u/\varepsilon$. The subscripts *i* and *o* stand for *inlet* and *outlet*, respectively. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.) Note:

$B(x) =$ elevation of bed bottom, m
$G(x) =$ elevation of bed surface, m
$H(x) =$ elevation of water surface, m
$P =$ precipitation, m/d
$x =$ distance from inlet, m
$ET =$ evapotranspiration, m/d
$h(x) =$ water depth, m
$L =$ bed length, m
$Q =$ volumetric flow rate, m^3/d
$\delta =$ bed depth, m

hydraulic conductivity are also present after the wetland has been in operation for a time. Most HSSF wetlands are rectangular, and so that feature is added to the list of restrictions. Notation is outlined on Figure 2.19.

The mass balances and geometrical definitions have been presented in Equations 2.20 through 2.22, which also hold for HSSF wetlands. The porosity is lower, usually in the range 0.35–0.45 m^3/m^3 for sands and gravels; and there is the added geometry of a bed surface to consider. The elevation of the top surface of the media is:

$$G = B + \delta \tag{2.38}$$

where
$G =$ elevation of the bed top above datum, m
$\delta =$ thickness of the bed media, m

The freeboard, or headspace, is defined to be the distance from the top surface of the media down to water:

$$f = \delta - h \tag{2.39}$$

where
$f =$ freeboard, m

In general, the variables h, H, G, δ, f, and B are each dependent on distance from the bed inlet.

Bed Friction and Hydraulic Conductivity

The simplest friction relationship states that superficial velocity is proportional to the slope of the water surface:

$$u = -k\frac{dH}{dx} \tag{2.40}$$

where
$H =$ elevation of the water surface, m
$k =$ hydraulic conductivity, m/d

This is the one dimensional version of Darcy's law. It is restricted to the laminar flow regime.

A more general correlation spans both laminar and turbulent flow. The laminar term in Equation 2.40 is preserved, and a turbulent term is added:

$$-\frac{dH}{dx} = \frac{1}{k}u + \omega u^2 \tag{2.41}$$

where
$\omega =$ turbulence factor, d^2/m^2

The turbulent contribution ωu^2 is negligible when the particle Reynolds number is less than 1.0, and may be ignored with small error at Reynolds numbers up to 10. The particle Reynolds number is defined as:

$$\mathrm{Re} = \frac{D\rho u}{(1-\varepsilon)\mu} \tag{2.42}$$

where
D = particle diameter, m
ρ = density of water, kg/m^3
μ = viscosity of water, kg/m·d

Sand media will typically be in the laminar range; but rock media will often be in the transition region between laminar and turbulent, with significant contributions from the turbulent term. Simple rearrangement of Equation 2.42 gives:

$$u = -k_e \frac{dH}{dx} \tag{2.43}$$

where
k_e = effective hydraulic conductivity, m/d

Comparison of Equations 2.41 and 2.43 indicates that:

$$\frac{1}{k_e} = \frac{1}{k} + \omega u \tag{2.44}$$

When velocity is beyond the laminar range, the effective hydraulic conductivity will depend on velocity.

Correlations for Hydraulic Conductivity of Clean Bed Porous Media

The original "clean bed" hydraulic conductivity and turbulence factor for a particulate media depend on the characteristics of the media:

1. Mean particle diameter
2. Variance of the particle size distribution
3. Particle shape
4. Porosity of the bed
5. Arrangement of the particles

Of these, the effects of particle size and porosity have been quantified in the form of equations in the nonwetland literature. For instance, the Ergun equation (Ergun, 1952) is widely accepted for random packing of uniform spheres:

$$-\frac{dH}{dx} = \left(\frac{150\left(1-\varepsilon\right)^2\mu}{\rho g \varepsilon^3 D^2}\right)u + \left(\frac{1.75\left(1-\varepsilon\right)}{g\varepsilon^3 D}\right)u^2 \tag{2.45}$$

where
H = elevation of water surface, m
ε = porosity, dimensionless
D = particle diameter, m
ρ = density of water, kg/m^3
μ = viscosity of water, kg/m/d
u = superficial flow velocity, m/d
g = acceleration of gravity, m/d^2

Comparison with Equation 2.41 indicates that:

$$k = \frac{\rho g \varepsilon^3 D^2}{150(1-\varepsilon)^2 \mu} \tag{2.46}$$

$$\omega = \frac{1.75(1-\varepsilon)}{g\varepsilon^3 D} \tag{2.47}$$

Equation 2.45 works for spheres of a single size; but gravel bed wetlands do not utilize such media. Hu (1992) applied Equation 2.45 to a HSSF system at Bainikeng, China, and found that Ergun-predicted depths were about 10 cm too large. The effects of a nonspherical shape are also significant (Brown and Associates, 1956). Idelchik (1986) gives a correlation for crushed, angular materials, which predicts conductivities about three times lower than those for spheres of the same size.

Most media possess a distribution of sizes. The presence of a particle size distribution lowers the hydraulic conductivity. This occurs because small particles have a disproportionately large amount of surface area, which causes drag on the water, and because the small particles can fit in the spaces between the larger particles. For instance, Freeze and Cherry (1979) present a technique based on work of Masch and Denny (1966) that utilizes the variance of the particle size distribution to estimate a correction factor for the hydraulic conductivity of large sand particles. For a variance of 50% of the mean particle size, the reduction is a factor of two.

Given all the uncertainties above, each of which can greatly influence the hydraulic conductivity of the clean media, it is prudent to measure the conductivity of the candidate media for a proposed project. Correlations may be used to guide the initial selection, but should not be trusted for final design purposes, because the gradient, porosity, and velocities have seldom been reported. Data for media from eighteen treatment wetland sites are displayed along with a prediction based upon a modification of Equation 2.45 in Figure 2.20. It is very important to recognize that Figure 2.20 is valid only for bare media with a porosity near 0.35 and a size variance near 50%.

Clogging of HSSF Bed Media

The HSSF bed will not maintain the clean-bed hydraulic conductivity once the system is placed into operation. For example, if one third of the pore space is blocked, the hydraulic conductivity will decrease by factor of ten, according to Equation 2.46, because hydraulic conductivity is extremely sensitive to porosity. This phenomenon must be acknowledged in design if the potential for bed flooding is to be minimized. Clogging of HSSF wetland beds occurs via the following mechanisms:

1. Deposition of inert (mineral) suspended solids in the inlet region of the wetland bed
2. Accumulation of refractory organic material (resistant to microbial degradation) in the inlet zone of the wetland bed

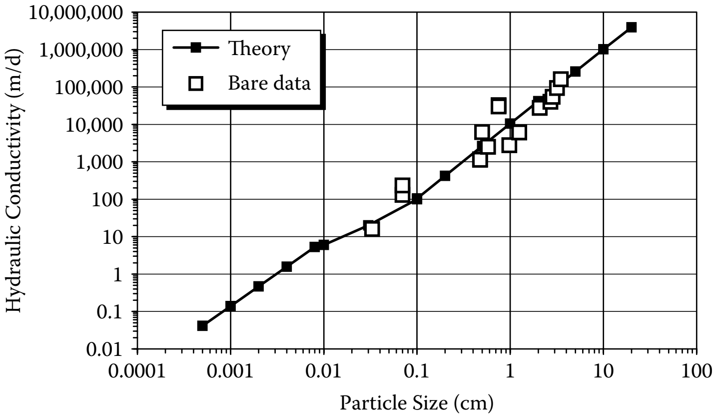


FIGURE 2.20 The dependence of clean bed hydraulic conductivity on media grain size. This plot is approximate; it is based on a porosity of 35%, and a 50% variance in the particle size distribution. Other size distributions, and deviation in particle shape and packing, will influence values for specific media. Data from 18 SSF wetlands is superimposed. (Data from Wolstenholme and Bayes (1990) In *Constructed Wetlands in Water Pollution Control*. Cooper and Findlater (Eds.), Pergamon Press, Oxford, United Kingdom, pp. 139–148; McIntyre and Riha (1991) *Journal of Environmental Quality* 20: 259–263; Sanford *et al.* (1995a) *Ecological Engineering* 4(4): 321–336; Kadlec and Watson (1993) In *Constructed Wetlands for Water Quality Improvement*. Moshiri (Ed.), Lewis Publishers, Boca Raton, Florida, pp. 227–235; Fisher (1990) In *Constructed Wetlands in Water Pollution Control*. Cooper and Findlater (Eds.), Pergamon Press, Oxford, United Kingdom, pp. 21–32; George *et al.* (1998) *Development of guidelines and design equations for subsurface flow constructed wetlands treating municipal wastewater.* Draft report to U.S. EPA, Cooperative Agreement CR818724–01–3, Cincinnati, Ohio; Watson and Choate (2001) *Hydraulic conductivity of onsite constructed wetlands*. Mancl (Ed.), Proceedings of the Ninth National Symposium on Individual and Small Community Sewage Systems; American Society of Agricultural Engineers: St. Joseph, Michigan, pp. 632–649; Drury and Mainzhausen (2000) *Hydraulic characteristics of subsurface flow wetlands*. Proceedings of the Billings Land Reclamation Symposium, Billings, Montana; graph from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

3. Deposition of chemical precipitates in the wetland bed
4. Loading of organic matter (both suspended and dissolved) that stimulates the growth of microbial biofilms on the bed media
5. Development of plant root networks that occupy pore volume within the wetland bed

Sediment Deposition

Solids deposition can occur for a variety of reasons, beginning with the placement of the media. Unwashed media will carry a load of fine dust or soil. Mud on the wheels of vehicles can add to the dirt supply during placement. And, those beds which are constructed with a layer of fine media on top of coarse media can be subject to the penetration of the lower layer by the upper-lying layer of finer material. Planting activities can introduce soils associated with the roots of the plants.

Due to the low flow velocities that occur within HSSF wetland beds, influent total suspended solids (TSS) will settle and deposit within the inlet region of the wetland bed. This deposition typically occurs within the first 5% of the wetland bed. As pore volume is occupied by suspended solids, the hydraulic conductivity is reduced accordingly, as described by Equation 2.46. This mechanism applies both to mineral (or inert) sediments as well as organic sediments that are refractory and resistant to microbial degradation (Mechanisms #1 and #2).

Chemical Precipitates

Chemical reactions within HSSF wetlands can result in the formation of insoluble chemical precipitates (Mechanism #3) (Liebowitz *et al.*, 2000; Younger *et al.*, 2002). These precipitates can also block pore spaces within the wetland bed and have the same effect in reducing hydraulic conductivity as described by Equation 2.46. Since the formation of precipitates is primarily governed by the redox potential within the wetland bed, reductions in hydraulic conductivity are not restricted to the inlet end of the wetland bed.

Biomat Formation

Microbial biofilms form in response to both particulate and soluble organic loading rates (Mechanism #4). These biofilms entrap both organic and inorganic solids (Winter and Goetz, 2003), forming a *biomat*. This biomat varies depending on the nature of the waste being treated. Biomat formation is greatest at the inlet end of the wetland where the organic loading is highest (Ragusa *et al.*, 2004). The loss of pore volume due to biomat formation reduces the hydraulic conductivity in this inlet zone (Zhao *et al.*, 2004). Organic matter is removed as wastewater flows through the wetland, resulting in declining biomat growth. At the outlet, where only small quantities of organic matter are available to microbes, biomat formation is negligible.

Plant Root Morphology

Wetland plants in HSSF systems develop a preferential rooting preference in the upper region of the granular bed (Mechanism #5). This rhizome morphology is strongly dependent on redox conditions within the HSSF bed (Lockhart, 1999) and is described further in Chapter 3. This limited root penetration can create preferential flow paths through the lower section of the gravel bed (Breen and Chick, 1995; U.S. EPA, 2000a; Whitney *et al.*, 2003; Nivala, 2005).

HSSF Bed Clogging in the Inlet Region

As a consequence of these factors, the HSSF bed will become clogged over time. Since the primary mechanisms for bed clogging predominate in the inlet region, the greatest reductions in hydraulic conductivity occur at the wetland inlet. Bed clogging has created the majority of operational problems for HSSF wetlands around the world. Bed clogging is not a new phenomenon, although the mechanisms by which it occurs are only now being elucidated. As noted by Zachritz and Fuller (1993), "Clogging ... has been an operational problem since plant start-up" at the Carville, Louisiana facility.

These operational problems are being evaluated in current HSSF wetlands. For instance, Cooper *et al.* (2006) report that 111 of 255 reedbeds inspected were flooded at the inlet end. These had a median age of about ten years.

Development of HSSF Bed Clogging

Although the processes of bed clogging are still being quantified, there appear to be two distinct sets of mechanisms that contribute to the problem:

- Short-term effects that reduce hydraulic conductivity over the first year of operation. These appear to be related primarily to the development of plant root networks (primarily in the upper regions of the wetland bed) and microbial biomat formation occurs primarily in the inlet region of the wetland (Mechanisms #4 and #5).
- Long-term effects that gradually reduce hydraulic conductivity. These appear to be primarily related to deposition of inert (mineral) suspended solids, accumulation of refractory organic material, and formation of insoluble chemical precipitates (Mechanisms #1, #2, and #3).

Short-Term Bed Clogging Mechanisms

The majority of porosity decrease appears to occur during the first year after bed commissioning (Figure 2.21). That is the period of principal root/rhizome grow-in, and the development of biofilms. In total, the losses in Figure 2.21 are 16% and 23%. The loss of porosity is reflected in a reduction of residence time. For example, Marsteiner *et al.* (1996) reported about a 10% loss of detention time for planted beds versus unplanted beds, and Tanner *et al.* (1998a) estimated root/rhizome blockage to be 4%.

The nonuniform distribution of roots and biomat along the length of the bed results in a nonuniform distribution of hydraulic conductivity throughout the bed, as shown schematically in Figure 2.22.

The result of porosity decrease is a severe decline in bed hydraulic conductivity, primarily in the front end of the bed. McIntyre and Riha (1991) showed that hydraulic conductivity drops during the early months of plant establishment (Figure 2.23). These mesocosms were fed a nutrient solution, and therefore incoming sediments were negligible. Interestingly, both planted and unplanted mesocosms showed reduced conductivity, with plants only slightly increasing the loss. Wolstenholme and Bayes (1990) documented a similar pattern of drastic conductivity reduction for four reedbeds at Valleyfield, Scotland, over the first year of operation.

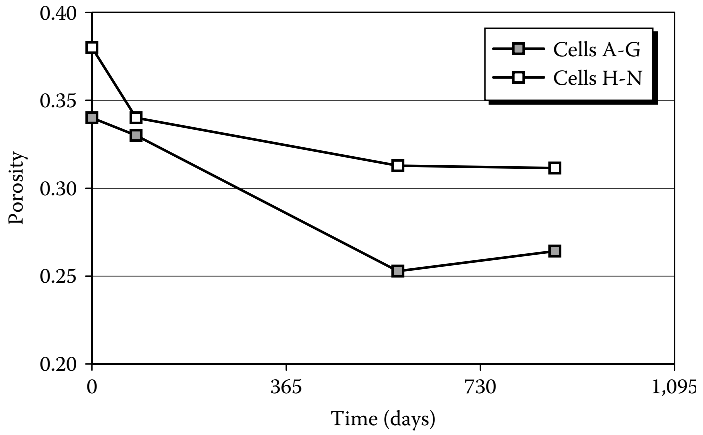


FIGURE 2.21 The decrease in HSSF bed porosity over time. (Data from George *et al.* (1998) *Development of guidelines and design equations for subsurface flow constructed wetlands treating municipal wastewater.* Draft report to U.S. EPA, Cooperative Agreement CR818724–01–3, Cincinnati, Ohio.)

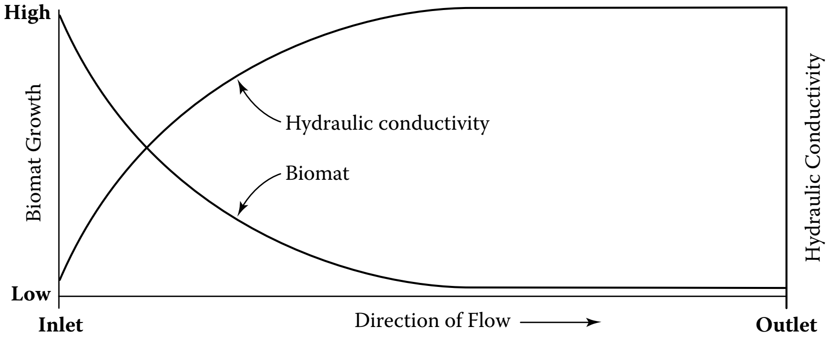


FIGURE 2.22 Relationship between hydraulic conductivity and biomat formation. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

Long-Term Bed Clogging Mechanisms

Regardless of short-term effects, HSSF wetlands that receive a sustained load of particulate matter will experience a continuing loss of porosity over time, and corresponding reductions in hydraulic conductivity. Accumulation of inert sediments and refractory organic material will occur primarily in the inlet region of the wetland bed, and exacerbate conductivity losses in this region. Formation of chemical precipitates is dependent on the redox conditions within the HSSF wetland bed, and may not be confined to just the inlet zone.

HSSF Bed Clogging

The combined effect of short-term and long-term bed clogging mechanism is to produce a drastic reduction in the hydraulic conductivity of the inlet zone of the wetland bed.

The magnitude of conductivity decline may well over a factor of ten, as it was at the Richmond, New South Wales, Australia, site (Figure 2.24). Most of the decline in this particular system was apparently associated with biomat formation in the inlet region, as unplanted gravel beds showed declines similar to those found in the planted systems (Fisher, 1990; Sanford *et al.*, 1994). The microbial populations associated with nutrient cycling and BOD reduction are highest in the inlet section of the bed, in response to the elevated contaminant concentrations in that region. These organisms, together with their associated biomats, reduce the pore volume in the entrance region of the bed to a greater extent than the downstream sections. In turn, this implies a greater reduction of hydraulic conductivity in the inlet region and a resulting nonuniform hydraulic gradient. This effect has been measured by several investigators (Fisher, 1990; Kadlec and Watson, 1993; Watson and Choate, 2001), as illustrated in Figure 2.25.

Kadlec and Watson (1993) found approximately 10% of the voids blocked by volatile and inorganic solids. Tanner *et al.* (1998a) investigated the loss of detention time (porosity) over a five year period for beds receiving organic solids loadings of 1.73, 2.09, and 5.80 g/m^2·d. Detention times were 101%, 61%, and 50% of theoretical, respectively. In consideration of these factors, there have been proposed relations

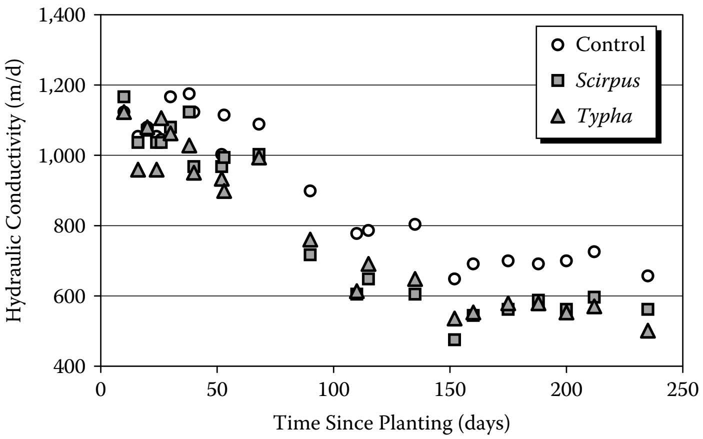


FIGURE 2.23 Decrease of hydraulic conductivity of planted laboratory mesocosms. There is evidence for reduction by both biofilms and root growth. (Data from McIntyre and Riha (1991) *Journal of Environmental Quality* **20**: 259–263.)

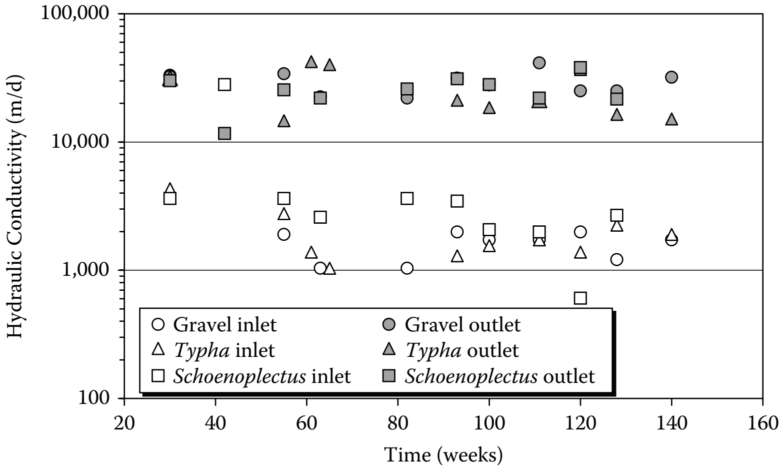


FIGURE 2.24 The dependence of hydraulic conductivity on time, adapted from Fisher (1990). Three beds are represented: an unplanted gravel bed, gravel planted with *Scirpus*, and gravel planted with *Typha*. Data fall into two groups: the inlet sections, 0–20 m from the inlet; and the rest of the bed, 20–100 m from the inlet. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

between time to complete clogging and the solids loading to the wetland (Bavor and Schulz, 1993; Blazejewski and Murat-Blazejewska, 1997; Langergraber *et al.*, 2003; Wallace and Knight, 2006). Additional factors that may contribute HSSF bed clogging are addressed in Chapter 7.

Soil based systems, as formerly used in Europe, may not display clogging problems, simply because the hydraulic conductivity assumed for design purposes (and associated inlet zone organic loadings) are already extremely low (ÖNORM B 2505, 1997; ATV, 1998) and, as a consequence, may match the conductivity of deposited influent solids. Haberl and Perfler (1990) found no evidence of reduced conductivities over five years for any of three HSSF soil-based wetlands at Mannersdorf, Germany. They found variable changes in hydraulic

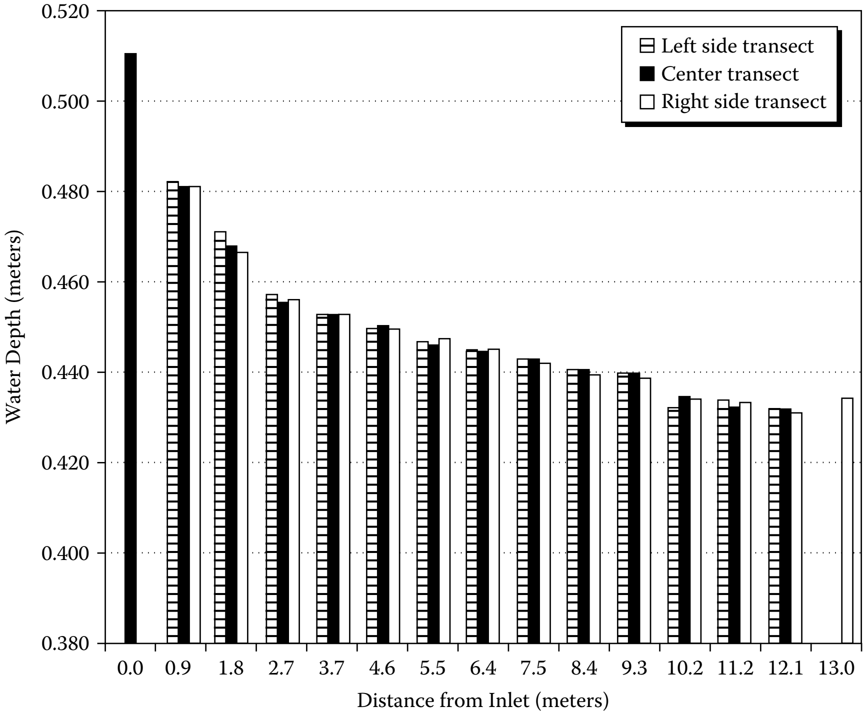


FIGURE 2.25 Water surface profile of a single-home HSSF wetland in Alabama. (From Watson and Choate (2001) *Hydraulic conductivity of onsite constructed wetlands*. Mancl (Ed.). Proceedings of the Ninth National Symposium on Individual and Small Community Sewage Systems; American Society of Agricultural Engineers: St. Joseph, Michigan, pp. 632–649. Reprinted with permission.)

TABLE 2.9
Typical Values of Hydraulic Conductivity for Soil Materials

Soil Texture	Hydraulic Conductivity (m/d)
Gravel, coarse sand	36.0 +
Coarse, medium sand	6.0–36.0
Fine sand, loamy sand	2.4–6.0
Sandy loam, loam	1.2–2.4
Loam, porous silt loam	0.6–1.2
Silty clay loam, clay loam	0.3–0.6

Source: Data from U.S. EPA (2002c) *Onsite wastewater treatment systems manual*, EPA 625/R-00/008, U.S. EPA Office of Research and Development: Washington, D.C.

conductivity, with minimum and maximum values of 0.37 ± 0.34, and 3.8 ± 2.2 m/d, respectively. Coombes (1990) reports conductivities for several United Kingdom HSSF reed beds, in the range of 0.2–9.8 m/d, which is the same as the range shown in Table 2.9.

The soil-based system at Acle, Norfolk, United Kingdom, had essentially the same conductivity at year four as at startup. However, because of low conductivity, soil-based systems are prone to flooding and commonly operate in a flooded mode (similar to a FWS wetland).

At the present time, there is not a clear understanding of the rate and development of HSSF bed clogging. Short-term mechanisms appear to be clearly related to plant root development, organic loading, and the size of the bed media. However, even if short-term clogging is avoided through the proper application of design principles, long-term clogging mechanisms are still operative. There is a growing body of knowledge in North America and Europe that indicates HSSF bed clogging may be inevitable (Cooper *et al.*, 2004; Wallace and Knight, 2006; Cooper *et al.*, 2006; Puigagut *et al.*, 2006). Whereas the rate of bed clogging is related to mechanisms discussed in this chapter, regularly scheduled replacement or cleaning of the bed media in the inlet zone of HSSF wetlands may be a routine (and unavoidable) part of the operation of HSSF wetlands.

The implications of these clogging phenomena in design are very important. The HSSF wetland must be able to operate properly and be capable of establishing and sustaining plant growth, in the face of large changes in hydraulic characteristics that will occur over the life of the system.

HSSF Water Elevation Profiles

More than just Darcy's law is required to calculate flow rates and depths in a HSSF wetland. Previous equations provide the ability to calculate h, H, u, and Q as functions of distance down the length of the bed. Because two of these are differential equations, an integration procedure must be implemented and boundary conditions must be specified. The required equations for the case of steady flow without atmospheric augmentation are:

$$\frac{d\left[u\,(H-B)\right]}{dx}=0 \tag{2.48}$$

$$-\frac{dH}{dx}=\frac{1}{k_e}u \tag{2.49}$$

Given a bottom elevation profile $B(x)$, Equations 2.48 and 2.49 provide the volume flow and depth profiles from:

$$Q=uWh \tag{2.50}$$

$$h=H-B \tag{2.51}$$

The boundary conditions would most often be a specification of the exit water elevation (set by a structure), and a specification of the inlet flow rate (set by the delivery system):

$$H_{at\ x=L}=H_o \quad \text{and} \quad Q_{at\ x=0}=Q_i \tag{2.52}$$

The required input information must also include: bed width W, the bottom elevation profile $B(x)$, the hydraulic conductivity profile $k(x)$, and the turbulence factor profile $\omega(x)$.

There is a very important constraint to be met in the course of solution of these model equations: flow must be underneath the media surface, or the hydraulic conductivity Equation 2.49 does not apply. Mathematically, this means satisfying the inequality:

$$0<h<\delta \tag{2.53}$$

These model equations may be solved on a spreadsheet without great difficulty.

Water Surface Calculations

The common assumption is that most HSSF beds have uniformly flat but inclined bottoms. Many operate in the laminar flow region. Since HSSF wetlands tend to be loaded at higher hydraulic loading rates than FWS wetlands, effects of atmospheric gains or losses are minimized. Most HSSF systems are intended to operate at constant water depth. In their initial startup condition, the hydraulic conductivity will not be a function of distance from the inlet. Under these ideal conditions, the model reduces to:

$$\frac{H_i-H_o}{L}=\frac{u_{avg}}{k}=\frac{1}{k}\frac{Q}{Wh_{avg}} \tag{2.54}$$

It is important to note that *the gradient on the left side of this equation is the slope of the water surface, not the bottom of the wetland bed.*

The bottom bed slope within a HSSF wetland will not drive the flow. Similar to lakes, if there is not a slope to the water surface, there is no flow of water.

The use of Equation 2.54 by itself for design can lead to serious errors, and has done so. The average water depth h_{avg}

is not adequate for design; the longitudinal depth profile is required. The problems are related to the large changes in k, to the violation of the constraint of Equation 2.53 (flooding), and to the sensitivity of vegetation to the headspace, $f = \delta - h$. Basically, the design goal is to keep the water below the surface of the bed, but high enough for plant roots to reach it.

Some sense of the validity and sensitivity of the model can be gained from its calibration to field data at Benton, Kentucky. The inlet zone of this crushed limestone HSSF cell was flooded at the time of the study, which was about three years after startup. Measurements included detailed surveys of the water surface elevation and of flow rates. The media were tested in the field to determine the *in situ* conductivity (TVA, 1989, unpublished data).

The washed media was also tested in the laboratory (Kadlec and Watson, 1993). Although the lab conductivity was roughly comparable to the field values, the lab value was clearly on the high side (67% above field mean). That kind of difference can easily arise from differences in void fraction resulting from packing factors. Further, the media in the bed contained some considerable amount of dirt from work vehicles. A twelve percent void fraction difference would account for the differences in k. The predicted profile of water depth is a straight line in Figure 2.26. However, the conductivity in the inlet zone was low, and increased markedly along the flow direction (Figure 2.27). If clogging is accounted for in the hydraulic conductivity profile, the model fits the water surface profile correctly (Figure 2.26).

These conditions created large variations from the original design intent. Effects on vegetation were huge due to hydroperiod shifts: the inlet section (20%) became an overland flow *Schoenoplectus* (*Scirpus*) wetland due to flooding. The remaining 80% of the bed contained only sparse terrestrial vegetation, because the plants responded to excessive headspace (f), which created a large unsaturated rooting depth and effectively precluded the growth of emergent wetland plants.

Flooded Operation

A combination of clogging and inappropriate design has produced overland flow in many existing HSSF systems (Figure 2.26). Flooding is usually confined to the inlet region of the bed. Overland flow carries the excess water until the hydraulic conductivity and flow gradient over the remaining travel distance are sufficient to permit the flow to be carried below ground. The amount of water carried in overland flow may be estimated from methods presented earlier in this chapter for FWS wetlands. If over-flooding depths are as much as a few centimeters, most of the water will be carried to the HSSF wetland outlet by overland flow. As reported by Spangler *et al.* (1976b), "*The main flow of water, however, was horizontally across the surface to the area immediately above the outlet, and then vertically down to the outlet....*" As a simple approximation, the HSSF system transforms into a FWS wetland.

The Denham Springs, Louisiana, HSSF wetlands are a case in point (Figure 2.28). Even though these beds are filled with 6 cm rock, a combination of the aspect ratio (L:W = 5:1) and the high flow (hydraulic loading rate of 25 cm/d) led to extensive inlet zone flooding. Predictions from Equation 2.54, even based on clean-bed hydraulic conductivity, indicate such flooding should be expected, but such calculations were apparently not performed.

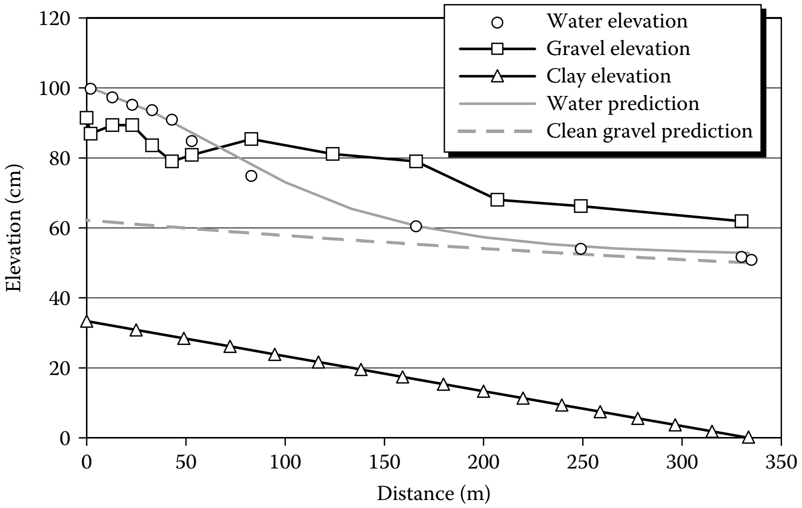


FIGURE 2.26 Water mounding due to clogging in a HSSF wetland. At the time of the study, the inlet section of the bed was clogged significantly. The hydraulic conductivity increased from 3,070 m/d in the inlet region, to 26,400 m/d in the outlet region. The clean gravel prediction is based on a laboratory measurement of k = 44,000 m/d. (Data from Kadlec and Watson (1993) In *Constructed Wetlands for Water Quality Improvement*. Moshiri (Ed.), Lewis Publishers, Boca Raton, Florida, pp. 227–235; graph from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

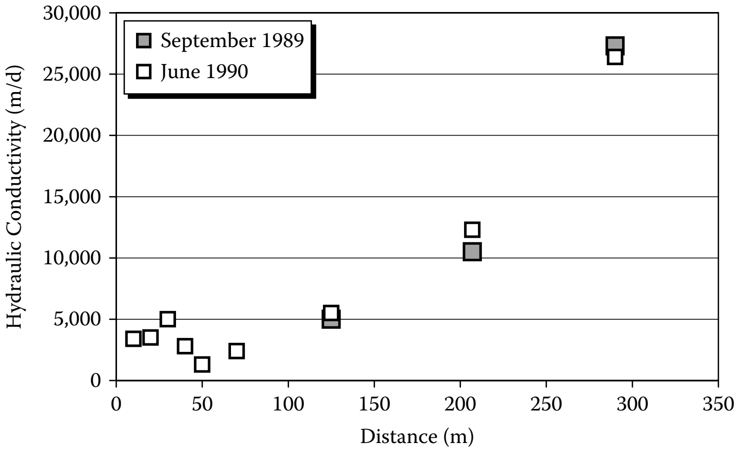


FIGURE 2.27 Hydraulic conductivity of the crushed stone media in operation two and three years at Benton, Kentucky. (From Kadlec and Watson (1993). In *Constructed Wetlands for Water Quality Improvement*. Moshiri (Ed.), Lewis Publishers, Boca Raton, Florida, pp. 227–235. Reprinted with permission.)

Flow Stratification

Several factors create vertical stratification of hydraulic conductivity in HSSF beds, and therefore lead to vertical profiles of water flow rates. There have been anecdotal reports of plant root penetration of up to 60 cm (Gersberg *et al.*, 1986), but several subsequent studies have found lesser rooting depths (Daniels and Parr, 1990; Parr, 1990; Pilgrim *et al.*, 1992; Adcock and Ganf, 1994; Kuusemets *et al.*, 2002). For

FIGURE 2.28 The flooded inlet zone of the Denham Springs, Louisiana, HSSF wetland.

Phragmites, in excess of 90% of the biomass of roots and rhizomes are in the upper 30 cm of the media (Figure 2.29). Other species follow similar patterns, but may have somewhat different mat thicknesses. For instance, *Schoenoplectus* (*Scirpus*) *validus* rooted to a thickness of about 20 cm at Benton, Kentucky (Figure 2.30); *Schoenoplecctus* (*Scirpus*) *sylvaticus* to 10 cm at Kodijärve, Estonia; *Typha* to 20 cm at Byron bay, Australia; and *Baumea articulata* to 30 cm at Adelaide, Australia. Laboratory evidence indicates that the depth of root penetration is strongly influenced by the redox potential within the wetland bed. Strongly reducing and nutrient-rich conditions are associated with shallower root penetration and less root biomass (Lockhart, 1999; Wallace and Knight, 2006). Since many HSSF wetlands are used in conjunction with primary-treated effluents, the resulting conditions in the wetland bed limit root penetration.

Cumulative experience with HSSF wetlands indicates that deeper gravel beds (>40 cm) will contain an upper zone that contains essentially all of the plant roots and a lower zone without roots. The presence of root blockage is an important factor: the root zone impedes flow more than the bare media below it. Several tracer studies have documented this phenomenon (Fisher, 1990; Pilgrim *et al.*, 1992; Marsteiner *et al.*, 1996; García, 2003).

There is considerable evidence that accumulated solids selectively occupy different layers in the media (Kadlec and Watson, 1993; Sanford *et al.*, 1995; Tanner *et al.*, 1998a). No common pattern has been found for the vertical profiles, with some studies showing more solids in the bottom (Kadlec and Watson, 1993; Tanner *et al.*, 1998a), while others show more solids near the surface (Tanner and Sukias, 1995). Deposition of solids within the HSSF bed is likely dependent on the suspended solids loading and the physical characteristics of the sediments (specific gravity, particle size, and biodegradability).

Finally, it has been found that density-induced stratification can occur in FWS wetlands (Stephan *et al.*, 2004) and also in HSSF beds (Rash and Liehr, 1999; Kadlec *et al.*, 2003).

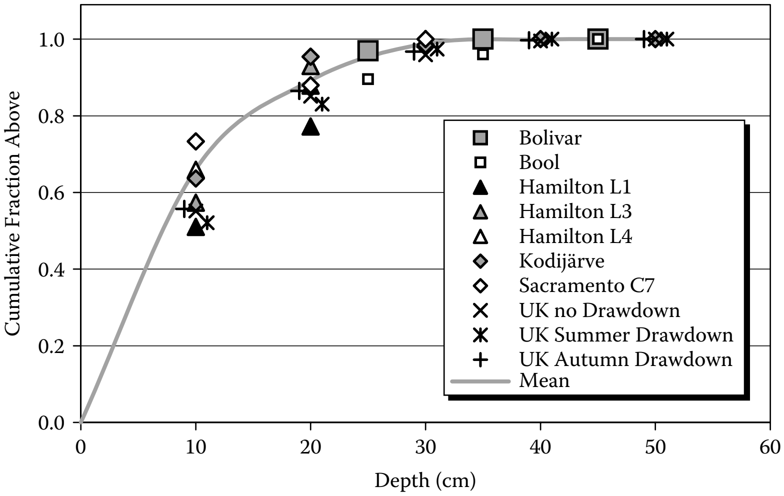


FIGURE 2.29 Depth profiles of roots and rhizomes for SSF Phragmites beds. (Data from Daniels and Parr (1990) *1989 Survey of reed growth in UK reed bed treatment systems*. WRc/NERC Contract Report T02058F1, The Institute of Terrestrial Ecology: Dorset, United Kingdom; Adcock and Ganf (1994) *Water Science and Technology* **29**(4): 95–102; Kuusemets ***et al.*** (2002) *Nitrogen and phosphorus assimilation and biomass production by* Scirpus sylvaticus *and* Phragmites australis *in a horizontal subsurface flow constructed wetland*. Mbwette (Ed.). Proceedings of the 8th International Conference on Wetland Systems for Water Pollution Control, 16–19 September 2002; Comprint International Limited: University of Dar Es Salaam, Tanzania, pp. 930–937.)

This phenomenon is caused by water of moderate salt content (higher density) moving along the wetland bottom, and rainwater of lesser salt content (lower density) moving in the upper strata of the bed. Such vertical density gradients are very stable, and persist for months (Kadlec *et al.*, 2003), perhaps indefinitely (Nivala, 2005) unless operational steps are taken to address the hydraulic short-circuiting that results from vertical stratification.

FIGURE 2.30 Roots of *Scirpus validus* in the Benton, Kentucky, HSSF wetland. Scale is in inches. The major portion of the root mat is contained in the top 20 cm.

2.4 VF WETLAND HYDRAULICS

Subsurface flow systems may be operated in vertical flow mode, but that mode has many variants. These include:

1. *Intermittent downflow.* This option involves flood application of water on top of the bed for brief periods of time. This operational mode is selected to enhance oxygen transport into the bed. This type is favored in many European countries. It was advanced as part of the original Max Planck Institute system developed in the 1960s (Seidel, 1966). When no plants are used, these are termed intermittent sand filters (Liénard *et al.*, 2001).
2. *Unsaturated downflow.* This variant involves distributing water across the top of a granular media. Water then trickles through the media in unsaturated flow. Distribution pipes may be located above the system, or, in cold climates, buried within the granular media bed. The system may be configured in a single-pass mode or, more commonly, employ flow recirculation so that wastewater passes through the media bed multiple times. These systems are functionally equivalent to recirculating sand or gravel filters (Crites and Tchobanoglous, 1998; Crites *et al.*, 2006). Systems with very low hydraulic loading rates may be unlined. If the primary intent is to harvest a crop, these systems are called slow-rate infiltration systems (Nichols and Boelter, 1982; Water Environment Federation, 2001).
3. *Saturated up- or downflow.* These systems employ continuous saturated flow of water through the

plant root zone. Downflow configurations are used in mine water treatment, where they are termed an anaerobic wetland or alkalinity producing system (Younger *et al.*, 2002). Aerated downflow systems have been employed as polishing reactors for removal of ammonia (Wallace *et al.*, 2006a). Saturated upflow is desirable when the daylighting water must be of highest quality, to minimize contact with contaminants, or root zone contact is to be maximized (Heritage *et al.*, 1995; Tanner *et al.*, 2002a). These systems have been employed in the laboratory as anaerobic reactors to provide reductive dehalogenation of chlorinated solvents (Pardue *et al.*, 2000; Kassenga *et al.*, 2003).

4. *Tidal flow (fill and drain).* These systems employ the cycling filling and draining of a granular bed. During the fill portion of the cycle, the wastewater is fed into the bottom of the wetland bed. Flow moves upwards, gradually filling the bed. Fill is complete when the surface is flooded. The pump is then stopped; the wastewater is then held in the bed in contact with the bacteria growing on the media. After a holding period, the wastewater is drained, and air enters the voids in the bed. These reactors create cycling redox conditions that contain both oxidizing and reducing phases (Maciolek and Austin, 2006). The fill and drain frequency depends on the application, but is typically about two hours in length (Sun *et al.*, 1999). Tidal flow wetlands may be run in parallel pairs, one filling while the other is draining. This mode has been termed reciprocating operation (Behrends, 2000). Flows are a combination of horizontal and vertical during filling, but mostly vertical downward during draining.

Other types of operation have also been tested; for instance, the subsurface introduction of wastewater into saline groundwaters. The density difference creates an upward buoyancy driving force, causing vertical upward flow (Watson and Rusch, 2001; Richardson *et al.*, 2004).

The basis for analysis of vertical flow systems is also the proposition of Darcy, developed in 1856 (Brown, 2002) for saturated flow. In combination with the water mass balance, it is easily extended to the more general case of unsaturated flow (Richards' equation, Freeze and Cherry, 1979). The one-dimensional dynamic version is:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x}\left[K\frac{\partial h}{\partial x}\right] \tag{2.55}$$

where

h = pressure head, m
K = *unsaturated* hydraulic conductivity, m/d
t = time, d
x = vertical distance, m
θ = fractional water content, m^3/m^3

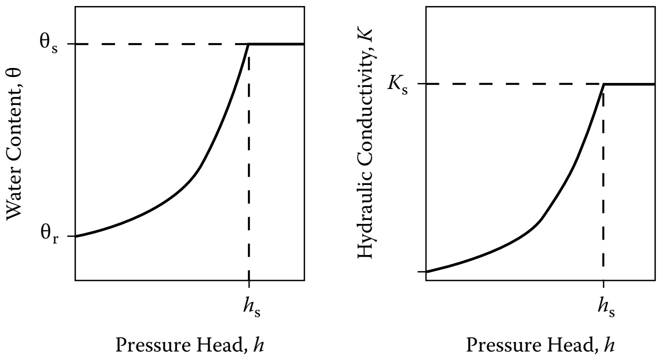


FIGURE 2.31 Both water content and hydraulic conductivity drop off markedly with decreasing pressure head for a typical porous substrate. At full capacity, the water content is θ_s, with the entire porosity filled. Some portion (θ_r) of the porosity is undrainable. In unsaturated conditions, the hydraulic conductivity decreases with decreasing water content, to only a small fraction of the saturated conductivity.

The complicating feature is that both the water content and the hydraulic conductivity are now functions of the pressure head:

$$\theta = \theta(h) \quad \text{and} \quad K = K(h) \tag{2.56}$$

These functional forms have been well-studied. In clean media, they are typically *s*-shaped curves (Figure 2.31). Dynamic computations of the flows and water contents under intermittent flow in vertical flow wetlands are possible, but require nontrivial numerical procedures (Schwager and Boller, 1997; Langergraber, 2001). Well-established computer programs are available for this purpose, such as HYDRUS-1D and HYDRUS-2D (Simunek *et al.*, 1998; 1999).

Here the results of these calculations are illustrated, to gain an appreciation of the course of a cycle of flooding and draining for a typical intermittent vertical downflow operation.

Intermittent Downflow Beds

Vertical flow beds often consist of layers of porous media, with the bottom-most layer consisting of coarse media with a network of perforated drainage pipes (Cooper *et al.*, 1996). An example of such layering used in Austria is shown in Figure 2.32 (ÖNORM B 2505, 1997). This bottom layer is freely draining, perhaps with a permanent pool level slightly above the bed bottom. At the continuous, low average hydraulic loading rates usually employed in treatment, a sand or gravel medium will not become saturated; and the flow will be percolation through voids partially filled with air. However, if the water is delivered in a short period of time, the instantaneous loading rate may exceed the drainage rate, and the media will then fill with water.

The sequence of events during a cycle of fast flooding and draining is conceptually straightforward. A dose of water to be treated is introduced as a flood on the bed surface, with up to six doses per day (Figure 2.33). A cycle begins with

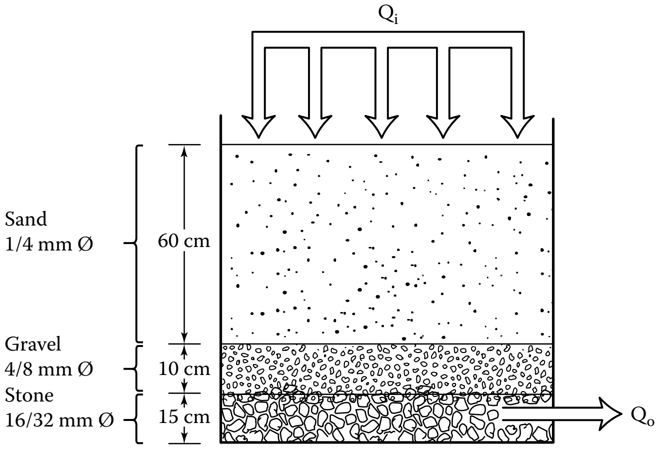


FIGURE 2.32 Vertical layering of an Austrian design for vertical flow constructed wetlands.

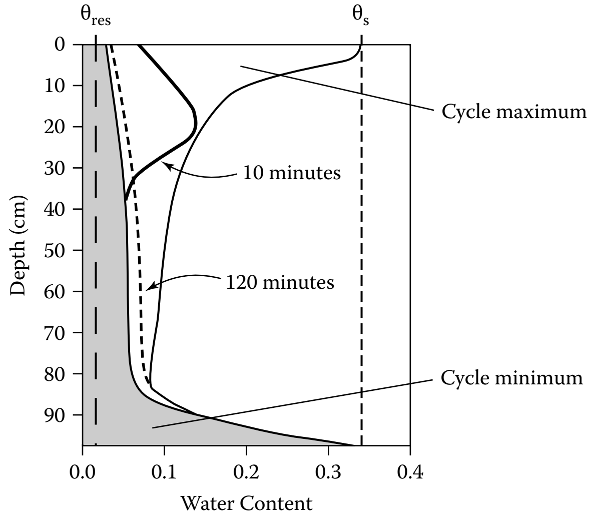


FIGURE 2.34 Water content profiles of a dosed vertical flow system. The cycle time was four hours (From Schwager and Boller (1997) *Water Science and Technology* 35(6): 13–20. Reprinted with permission.)

a mostly drained bed, containing water at or just above the residual water content of the media (Figure 2.34). The duration of the water introduction is variable, ranging from less than ten minutes (Schwager and Boller, 1997) to an hour or two (Watson and Danzig, 1993). For illustration, suppose the flood is very brief. The effect is then to create saturated conditions in the top layer of the bed, and perhaps surface ponding. The air in the voids is then trapped, and may be compressed by the water above if there is no air relief mechanism in the bed, such as a vent pipe. The period of air entrapment is typically brief, but may last for up to 45 minutes for large dosing pond depths (Schwager and Boller, 1997). Air bubbles can form, venting a portion of the trapped antecedent air (Figure 2.35).

After the air-lock is broken, drainage proceeds as unsaturated flow. Air enters the pores on top of the bed to replace the draining water volume in the bed voids. Air movement in the lower portions of the bed is minimal. Drainage in many VF systems is complete well before the start of the next cycle (Figure 2.34). During the fully drained portion of the period, air moves into the voids as determined by oxygen consumption and diffusion. Thus it is seen that the air deep in the bed has been convected to that location, while air in the upper layers of the bed has been supplied by diffusion (Kayser and Kunst, 2005).

The result of this cyclic operation is a variable outflow from the system. There is a rising outflow for a brief period, followed by a declining outflow (Watson and Danzig, 1993; Langergraber, 2001; Kayser and Kunst, 2005; Dittmer *et al.*, 2005) (Figure 2.36).

A period of resting, after full drainage, is typically included to allow for the oxidation of accumulated organics in and on the top of the bed, to avoid clogging.

FIGURE 2.33 The flooding of the vertical intermittent sand filters at the Saginaw, Michigan, landfill leachate treatment system. Water is dosed twice per day, with each flooding lasting only a few minutes. Rip-rap splash pads prevent erosion for the three-point distribution.

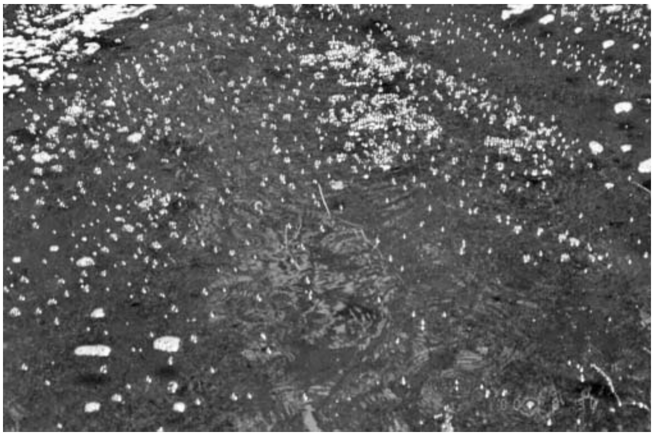


FIGURE 2.35 Trapped air bubbles up through the ponded water just after dosing at the Saginaw, Michigan, site.

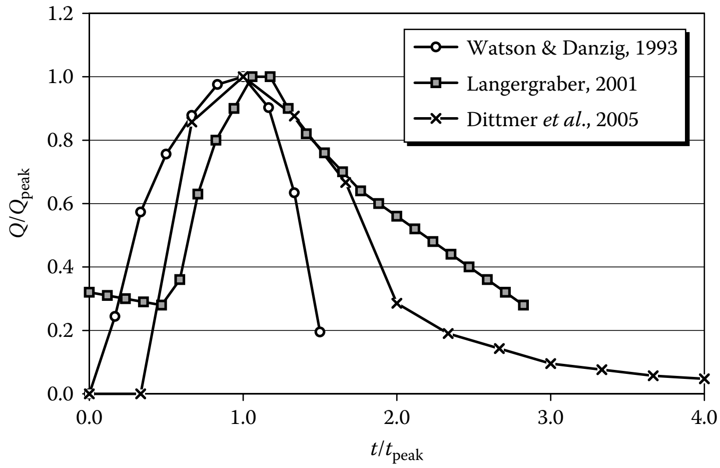


FIGURE 2.36 Outflow time series for vertical flow, dosed wetlands. (Data from Watson and Danzig (1993). In *Constructed Wetlands for Water Quality Improvement*, Moshiri (Ed.), CRC Press, Boca Raton, Florida, pp. 301–313; Langergraber (2001) Development of a simulation tool for subsurface flow constructed wetlands. Ph.D. Dissertation, IWGA SIG University of Vienna; Dittmer *et al.* (2005) *Water Science and Technology* 51(9): 225–232.)

There are many variants on this simple scheme. For instance, Green *et al.* (1997a; 1998) proposed that water should be accumulated in the bottom of the bed, and discharged rapidly, to induce air entry during the sudden drainage. During the refilling portion of the cycle, "used" air is vented through a perforated pipe located in the media.

Brix and Schierup (1990) provide explanation of a long-term dose and drain hydraulic operation. In their example, the bed is loaded at a constant rate for two days, followed by eight days of draining. The conditions for ponding may be expressed in terms of hydraulic loading:

$$q \geq k \tag{2.57}$$

where
k = saturated hydraulic conductivity of the media, m/d
q = instantaneous hydraulic loading rate, m/d

At lower loading rates, $q \leq K$, the bed can transport all water under the influence of the vertical pressure gradient of a fully saturated bed. During the relatively long period of water addition, the ponding depth increases exponentially, as does the drainage rate. After the cessation of water addition, the ponded water drains, after which the interstitial water drains. Depending on the hydraulic conductivity of the bed, drainage takes 0.5–10 days.

This same long-cycle operation was adopted in the project described by Kadlec *et al.* (1997) and Burgoon *et al.* (1999), only with loadings less than the hydraulic conductivity of the bed. Therefore, a period of unsaturated vertical flow was followed by a period of some days of draining and resting. Morris (1999) dosed intermittently at 3–10 near-instantaneous doses per day for two days, followed by six days' resting.

Empirical equations have been developed to describe the infiltration rates for various media and organic layers for systems designed and operated according to French criteria (Molle *et al.*, 2006). These are heavily loaded, dosed vertical beds. However, site conditions and operating strategies are quite variable, and it is thus not feasible to develop such descriptions for a general case.

Vertical Flow Tracer Tests

When an inert tracer is added to an intermittently dosed system, the response curves follow the general bell-shaped form that is seen for continuous flow systems, but with slight deflections during the course of each cycle (Schwager and Boller, 1997; Tanner *et al.*, 2002a). For intermittent vertical downward flows, the results of Schwager and Boller (1997) showed a gamma function response, with $N = 5$, and a tracer detention time of about 18 hours. Dosing was every four hours, and the average loading was 12 cm/d on a 90-cm-deep bed. The saturated water volume was about 35%, for a nominal saturated detention time of 2.5 days. However, the bed was far from saturated throughout most of flow (see Figure 2.34), and thus the detention time was much less than for saturated flow. Interestingly, the effect of clogging over time in this mode of operation was to cause an increase in detention time. This was due to the increased water holdup in the "used" bed, although the bed was still not saturated during flow for the clogged condition.

The vertical upflow, saturated system of Tanner *et al.* (2002a) provided an entirely different hydraulic environment. Water was displaced upward in each of five tanks in series, with the overflow from each being piped to the bottom of the next. Dosing was on a six-hour schedule, with about 2 L per dose. Each tank had a clean pore volume of about 8 L, thus the detention time was nominally five days. Four such cascades were operated, two with high-strength meat and dairy wastewaters, and two with pretreated dairy waters. The tracer detention time distributions all showed gamma distributions, with approximately two "tanks" per each of the five cells in series. However, the two cascades with strong influents displayed (measured) pore blockage of 46–64%, while the pretreated waters caused only half that amount of blockage. This reduction in pore volume caused the detention times to be shortened considerably compared to the nominal clean-volume calculated detention times.

Rogers *et al.* (1990) performed tracer tests on mesocosms in both an upflow and a downflow mode of operation, both batch-dosed twice per day at a hydraulic loading of 2.5 cm/d. The media was fully saturated at all times, with outflow taken as an overflow. They found major differences in root distribution, and concluded that the downflow mode was more akin to plug flow because of a surficial root mat.

Clogging

Clogging is a well-known phenomenon in sand filtration (Woodward and Ta, 1988). Platzer and Mauch (1997) conducted a literature survey in the context of vertical flow wetlands, and identified three potential mechanisms, to which one may add the presence of roots (Winter and Goetz, 2003):

1. Deposition and filtration of incoming particulates, leading to blockage of pores, especially near the surface
2. Biomass production in the soil pores, otherwise known as biomat formation, due to the favorable conditions created by domestic or municipal wastewater
3. Chemical precipitation in the pores, for example, calcium carbonate
4. The presence of roots

The third mechanism may be of concern in mine water treatment, but would otherwise not be expected to contribute. Inorganic materials accumulated via Mechanisms #1 and #2 are expected to remain in the pores, and eventually create blockage. However, organic materials are subject to oxidation, especially during the resting portion of a cycle, and are therefore removed at some speed determined by decomposition processes. Smaller grain sizes contribute to more rapid clogging. Roots and biofilms are anticipated to block only a small fraction of the pores (Langergraber *et al.*, 2003). We are then left with accumulation of solids as the principal mechanism of clogging.

The Effect of TSS on Clogging

As a first approximation, solids accumulate in pores as the result of complete filtration of incoming water:

$$M = qC_i \tag{2.58}$$

where
C_i = inlet TSS concentration, g/m^3
M = mass accumulation rate in pores, g/m^2·d
q = hydraulic loading rate, m/d

The result of this blockage is to increase the headloss required to drive the (constant) flow q through the bed. In constant flow potable water treatment, the head loss is inversely proportional to the remaining free area for flow (Woodward and Ta, 1988):

$$L = \frac{L_o}{1 - C_i M t} \tag{2.59}$$

where
C_i = inlet TSS concentration, g/m^3
L = headloss at time t, m
L_o = headloss at time zero, m
t = time, days

This simple relation does a creditable job of explaining operation of potable water sand filters (Woodward and Ta, 1988), and the concept has been adopted for vertical flow wetlands by Blazejewski and Murat-Blazejewska (1997), Langergraber *et al.* (2003), and Zhao *et al.* (2004). Of interest for the wetland application is the time to clogging, which is determined as a volume of accumulated solids:

$$t_{clog} = a\,\frac{\rho_{solid}}{qC_i} \tag{2.60}$$

where
a = empirical coefficient, m
t_{clog} = clogging time, days
ρ_{solid} = bulk density of accumulating solids, kg/m^3

Langergraber *et al.* (2003) suggest a = 0.18 m. Their lab data strongly support Equation 2.60. Blazejewski and Murat-Blazejewska (1997) propose:

$$a = 150\varepsilon \cdot d \tag{2.61}$$

where
d = particle diameter, m
ε = clean porosity, m^3/m^3

Equation 2.61 is based upon a clogging depth that is proportional to particle diameter. A good fit to the data of Bavor and Schulz (1993) was found. For very strong wastes, the clogging times, both real and predicted by Equation 2.60, are quite short — a matter of a few days (Zhao *et al.*, 2004).

The value of the coefficient a is very likely to depend upon the size distribution of particles and other bed properties. However, these studies have verified that Equation 2.60 provides a reasonable approximation to clogging due to filtration. The results from a number of VF wetlands suggest a sustained TSS loading of about 5 g/m^2·d is all that can be tolerated, but if there is adequate recovery in resting periods, then much higher rates can be sustained.

The Effect of Organic Content on Clogging

One concept of clogging argues that stronger influents (more BOD or COD) should promote more biofilm growth within the bed, and hence contribute to clogging. Blazejewski and

Murat-Blazejewska (1997) assume that biofilm growth and decomposition are in balance and do not contribute to clogging. Langergraber *et al.* (2003) conclude that biomass growth plays only a minor role compared to suspended solids over the short term. Winter and Goetz (2003) note that TSS and COD are often strongly correlated in the water to be treated, and that it is therefore difficult to sort out the possibilities.

The idea of organic materials contributing a major amount of blockage finds strong support when resting periods are included in the cycle. As noted by Platzer and Mauch (1997), the original conductivity of a bed may often be restored by allowing several days rest. Presumptively, this aerobic resting period causes oxidation of organics, thus freeing pore volume again. Platzer and Mauch (1997) reported a linear decrease in bed conductivity with increasing COD loading, but it is likely that TSS loading also increased. As a result of these uncertainties, a maximum COD loading has been deemed prudent. Platzer and Mauch (1997) suggest 25 $g/m^2 \cdot d$, and Winter and Goetz (2003) suggest 20 $g/m^2 \cdot d$.

Interestingly, there appears to be a second viable operating range, with very high TSS and COD loadings (Molle *et al.*, 2006). The TSS forms a mat on top of the bed, to depths in excess of 20 cm of organic material. This apparently acts as a trap for most incoming TSS, and spares the underlying bed from clogging. Up to 250 $g/m^2 \cdot d$ of COD (30–60 $g/m^2 \cdot d$ of BOD), and 20–50 $g/m^2 \cdot d$ TSS, have been sustainably treated (Chazarenc and Merlin, 2005). Accumulated solids form a compost layer on top of the original bed, with amounts of 20–90 kg/m^2 accruing after several years (Chazarenc and Merlin, 2005). This accretion is an effective mulch layer, which also aids in treatment. The surface water ponding that accompanies this mode of operation is problematical for single-home onsite treatment systems in North America, as these septic system codes typically require no daylighting of raw wastewater.

SUMMARY

This chapter has presented a synthesis of tools necessary to predict water budgets and hydraulics in treatment wetlands. Hydraulic processes in FWS and HSSF wetlands are similar in many respects, but there are significant differences. Adequate prediction methods are critical for treatment wetlands design and successful operation.

Wetland water budgets are dominated by surface inflows and outflows, evapotranspiration, and precipitation. Groundwater interactions are normally slight. Surface flows are generally measurable with sufficient precision. Precipitation may be projected from historical weather data, with the possibility of some error due to changing climatic conditions. Stochastic variability is large, however, on several times scales of interest. Atmospheric additions and losses are predictable by several techniques described in this chapter. In lightly loaded wetlands in warm seasons, this contribution may be very important in design calculations, so methods are presented for modifying pollutant reduction computations.

The internal water budget, or mass balance, for a treatment wetland is required for both conveyance calculations and pollutant reduction models. These equations, which have been detailed for FWS and HSSF wetlands, allow calculations of water depths and elevations and flow rates at interior points in the treatment wetland. Head losses in FWS wetlands have sometimes caused operational problems, and have often caused such difficulties in HSSF systems. Procedures for estimating frictional effects in both types of wetlands have been presented, along with shortcut methods for estimating the necessary design parameters to ensure adequate conveyance.

VF wetlands have been operated in both saturated and unsaturated flow regimes. For vertical saturated flow, many of the concepts outlined for HSSF wetlands directly apply. Pulse loading and associated unsaturated flow has also been discussed in this chapter.

3 Treatment Wetland Vegetation

There are many general functions of vegetation in wetlands. Physical functions include transpiration, flow resistance, and particulate trapping, all of which are related to vegetation type and density. Ecological functions include wildlife habitat and human use values. The focus here is water quality and, in particular, the processing of potential pollutants.

There are many effects vegetation can have on chemical processing and removal in treatment wetlands. These may include:

1. The plant growth cycle seasonally stores and releases nutrients, thus providing a "flywheel" effect for a nutrient removal time series.
2. The creation of new, stable residuals accrete in the wetland. These residuals contain chemicals as part of their structure or in absorbed form, and hence accretion represents a burial process for nitrogen.
3. Submersed litter and stems provide surfaces on which microbes reside. These include nitrifiers and denitrifiers, and other microbes that contribute to chemical processing.
4. The presence of vegetation influences the supply of oxygen to the water. Emergent vegetation blocks the wind, and shades out algae, presumably lowering reaeration. Floating vegetation may provide a barrier to atmospheric oxygen transfer. Submerged vegetation may provide photosynthetic oxygen supply directly in the water. To some limited extent, plant oxygen flux supplies protective oxidation in the immediate vicinity of plant roots.
5. The carbon content of plant litter supplies the energy need for heterotrophic denitrifiers.

Plants that occur in natural wetlands are described in many guidebooks and reference collections. They may be categorized by their growth habit with respect to the wetland water surface as:

- Emergent soft tissue plants
- Emergent woody plants
- Submersed aquatic plants
- Floating plants
- Floating mats

Obviously, only the first two categories may be implemented in SSF wetlands, whereas all five are candidates for FWS systems. The emphasis of treatment wetland technology to date has been on soft tissue emergents, including *Phragmites*, *Typha,* and *Schoenoplectus* (*Scirpus*).

Plant selection and establishment for constructed wetlands is covered in Chapters 18 and 21. The topic of biodiversity is covered in Chapter 19. In this chapter, plant species and examples of their usage are described. It is not the intent to provide full botanical specifications, but rather to acquaint the reader with the wide variety of choices of vegetation that have been implemented, and the sources of information that form the botanical foundation of treatment wetlands.

Because of the presence of ample water, wetlands are typically home to a variety of microbial and plant species. The diversity of physical and chemical niches present in wetlands results in a continuum of life forms from the smallest viruses to the largest trees. This biological diversity creates interspecific interactions, resulting in greater diversity, more complete utilization of energy inflows, and ultimately to the treatment properties of the wetlands ecosystem.

The study of organisms and their populations is a convenient way to catalog these life forms into groups with general similarities. However, the genetic and functional responses of wetland organisms are essentially limitless and result in the ability of natural systems to adapt to changing environmental conditions such as the addition of wastewaters. Genetic diversity and functional adaptation allow living organisms to use the constituents in wastewaters for their growth and reproduction. In using these constituents, wetland organisms mediate physical, chemical, and biological transformations of pollutants and modify water quality. In wetlands engineered for water treatment, design is based on the sustainable functions of organisms that provide the desired transformations.

The wetland treatment system designer should not expect to maintain a system with just a few known species. Such attempts frequently fail because of the natural diversity of competitive species and the resulting high management cost associated with eliminating competition, or because of imprecise knowledge of all the physical and chemical requirements of even a few species. Rather, the successful wetland designer creates the gross environmental conditions suitable for groups or guilds of species; seeds the wetland with diversity by planting multiple species, using soil seed banks and inoculating from other similar wetlands; and then uses a minimum of external control to guide wetland development. This form of ecological engineering results in lower initial cost, lower operation and maintenance costs, and most consistent system performance.

This chapter presents an overview of the floristic diversity that naturally develops in treatment wetlands as well as some details of the community types that may be fostered in wetland treatment systems. These microbial and plant species are typically the dominant structural and functional components in treatment wetlands. An understanding of their basic ecology will provide the wetland design or operator with insight into the mechanics of their "green" wastewater treatment unit.

Information about wetland plant species is voluminous and available from multiple sources. For more detailed information on aquatic and wetland microbial communities the reader is referred to Portier and Palmer (1989), Pennak (1978), or Wetzel (2001). For more detailed information on the ecology of the vascular plant species found in wetlands, the reader is referred to Hutchinson (1975), Sainty and Jacobs (1981), Brock *et al.* (1994), Reddington (1994), Cook (1996; 2004), Mitsch and Gosselink (2000a), or Cronk and Fennessy (2001). There are also multiple regional guides for the nonbotanist, for instance, for the northern United States:

Through the Looking Glass: A Field Guide to the Aquatic Plants. S. Borman, R. Korth, and J. Temte, 1997. Wisconsin Department of Natural Resources Publication No. FH-207-97, University of Wisconsin Extension, Stevens Point, Wisconsin.

National List of Plant Species That Occur in Wetlands for USFWS Region 3 (MI, IN, IL, MO, IA, WI, MN), A Field Guide. Resource Management Group, Inc., 1992. Prepared by Resource Management Group, Inc., Grand Haven, Michigan.

A Naturalist's Guide to Wetland Plants: An Ecology for Eastern North America. D.D. Cox, 2002. Syracuse University Press, Syracuse, New York.

A Field Guide to Wetland Characterization and Wetland Plant Guide: A Non-Technical Approach. K. Pritchard, 1991. Washington State University, Cooperative Extension Service, Seattle, Washington.

As another example source, the University of Florida Institute of Food and Agricultural Services maintains the Aquatic, Wetland, and Invasive Plant Information and Retrieval System (APIRS). Available are videos, line drawings, identification decks of color photos, and searches of a 50,000-record database (http://plants.ifas.ufl.edu). Thus, the practitioner can easily find scientific and common names, and gain an appreciation for what the plant looks like and its habitat requirements. We are therefore not reproducing this information here.

3.1 ECOLOGY OF WETLAND FLORA

Wetland Bacteria and Fungi

Wetland and aquatic habitats provide suitable environmental conditions for the growth and reproduction of microscopic organisms. Two important groups of these microbial organisms are bacteria and fungi. These organisms are important in wetland treatment systems primarily because of their role in the assimilation, transformation, and recycling of chemical constituents present in various wastewaters. Bacteria and fungi are typically the first organisms to colonize and begin the sequential decomposition of solids in wastewaters (Gaur *et al.*, 1992). Also, microbes typically have first access to dissolved constituents in wastewater and either accomplish sorption and transformation of these constituents directly or live symbiotically with other plants and animals by capturing dissolved elements and making them accessible to their symbionts or hosts.

The taxonomy of microbes is complex and frequently revised, but the general groups of bacteria and fungi are commonly recognized. Bacteria are classified in the Procaryotae (Buchanan and Gibbons, 1974). Procaryotes are distinguished by their lack of a defined nucleus with nucleaic material present in the cytoplasm in a nuclear region. Cyanobacteria or blue-green algae are also classified as procaryotes, but they are discussed with algae below. Fungi are classified as eucaryotes because they have a nucleus separated from the cytoplasm by a nuclear membrane.

Bacteria

Bacteria are unicellular, procaryotic organisms classified by their morphology, chemical staining characteristics, nutrition, and metabolism. *Bergey's Manual* (Buchanan and Gibbons, 1974) places bacteria into 19 associated groups with unclear evolutionary relationships. Most bacteria can be classified into four morphological shapes: coccoid or spherical, bacillus or rodlike, spirillum or spiral, and filamentous. These organisms may grow singly or in associated groups of cells including pairs, chains, and colonies. Bacteria typically reproduce by binary fission, in which cells divide into two equal daughter cells. Most bacteria are heterotrophic, which means they obtain their nutrition and energy requirements for growth from organic compounds. In addition, some autotrophic bacteria synthesize organic molecules from inorganic carbon (carbon dioxide, CO_2). Some bacteria are sessile while others are motile by use of flagella. In wetlands, most bacteria are associated with solid surfaces of plants, decaying organic matter, and soils.

Fungi

Fungi represent a separate kingdom of eucaryotic organisms and include yeasts, molds, and fleshy fungi. All fungi are heterotrophic and obtain their energy and carbon requirements from organic matter. Most fungal nutrition is saprophytic, which means it is based on the degradation of dead organic matter. Fungi are abundant in wetland environments and play an important role in water quality treatment. For general information about fungi, see Ainesworth *et al.* (1973).

Fungi are ecologically important in wetlands because they mediate a significant proportion of the recycling of carbon and other nutrients in wetland and aquatic environments. Aquatic fungi typically colonize niches on decaying vegetation made available following completion of bacterial use. Saprophytic fungal growth conditions dead organic matter for ingestion and further degradation by larger consumers.

Fungi live symbiotically with species of algae (lichens) and higher plants (mycorhizzae), increasing their host's efficiency for sorption of nutrients from air, water, and soil. If fungi are inhibited through the action of toxic metals and other chemicals in the wetland environment, nutrient cycling of scarce nutrients may be reduced, greatly limiting primary productivity of algae and higher plants. In wetlands, fungi are typically found growing in association with dead and decaying plant litter.

Microbial Metabolism

Microbes are involved in a large proportion of wetland transformations and removals. In many cases, there are several interconnected steps and organisms. The reader is referred to Maier *et al.* (2000) for an introduction to environmental microbial processes. Most of the important chemical transformations conducted by microbes are controlled by enzymes, genetically-specific proteins that catalyze chemical reactions. To a varying extent, bacteria and fungi are classified by their ability to catalyze certain reactions. Microbial metabolism includes the use of enzymes to break apart complex organic compounds into simpler compounds with the release of energy (catabolism) or the synthesis of organic compounds (anabolism) by the use of chemically stored energy. Microbial metabolism not only depends on the presence of appropriate enzymes but also on environmental conditions such as temperature, dissolved oxygen (DO), and hydrogen ion concentration (pH). Also, the concentration of the chemical substrate undergoing the transformation is of primary importance in determining reaction rate.

Microbes can be classified by their metabolic requirements. Photoautotrophic bacteria such as the green and purple sulfur bacteria use light as an energy source to synthesize organic compounds from CO_2. Reduced sulfur compounds such as hydrogen sulfide or elemental sulfur serve as electron acceptors in oxidation-reduction reactions. Photoheterotrophs use light as an energy source and organic carbon as a carbon source for cell synthesis. The organic carbon sources most typically used by photoheterotrophs are alcohols, fatty acids, other organic acids, and carbohydrates. Because photosynthetic bacteria do not use water to reduce CO_2, they do not produce O_2 as a byproduct of metabolism, as do the algae and higher plants.

Chemoautotrophic bacteria derive their energy from the oxidation of reduced inorganic chemicals and use CO_2 as a source of carbon for cell synthesis. A number of the bacteria which are important in wetland treatment of wastewater are chemoautotrophs. Bacteria in the genus *Nitrosomonas* oxidize ammonia nitrogen to nitrite, and *Nitrobacter* oxidize nitrite to nitrate, deriving energy, which is used in cell metabolism (see Chapter 9). The genus *Beggiatoa* derives energy from the oxidation of H_2S, *Thiobacillus* oxidizes elemental sulfur and ferrous iron, and *Pseudomonas* oxidizes hydrogen gas (see Chapter 11). Chemoheterotrophs derive energy from organic compounds and also use the same or other organic compounds for cell synthesis. Most bacteria, and all fungi, protozoans, and higher animals are chemoheterotrophs.

During microbial metabolism, carbohydrates are broken into pyruvic acid with the net production of two pyruvic acid molecules and two adenosine triphosphate (ATP) molecules for each molecule of glucose and the subsequent decomposition of pyruvic acid through fermentation or respiration. Fermentation by substrate-level phosphorylation does not require oxygen and results in the formation of a variety of organic end products such as lactic acid, ethanol, and other organic acids.

Aerobic respiration is the process of biochemical reactions by which carbohydrates are decomposed to CO_2, water, and energy (38 ATP molecules for each glucose molecule fully oxidized). The Krebs Cycle results in the loss of carbon dioxide (decarboxylation) and energy storage (two molecules of ATP per molecule of glucose). For complete oxidation to occur, oxygen and hydrogen ions must be available as the final electron acceptor in a chain of reactions called the electron transport chain. The overall reaction for aerobic respiration can be summarized as follows:

$$\begin{aligned} C_6H_{12}O_6 + 6H_2O + 6O_2 + 38\ \mathrm{ADP} + 38\ \mathrm{P} \\ = 6CO_2 + 12H_2O + 38\ \mathrm{ATP} \end{aligned} \tag{3.1}$$

Also, approximately 60% of the energy of the original glucose molecule is lost as heat during the complete aerobic respiration process.

Anaerobic respiration is an alternative catabolic process that occurs in the absence of free oxygen gas. In anaerobic respiration, some other inorganic compound is used as the final electron acceptor. A variable and lower amount of energy is derived during the process of anaerobic respiration. This form of respiration is important to several groups of bacteria which occur in wetlands and aquatic habitats. Bacteria in the genera *Pseudomonas* and *Bacillus* use nitrate nitrogen as the final electron acceptor, producing nitrite, nitrous oxide (N_2O), or nitrogen gas (N_2) by the process termed denitrification. *Desulfovibrio* bacteria use sulfate (SO_4^{2-}) as the final electron acceptor resulting in the formation of H_2S. *Methanobacterium* uses carbonate (CO_3^{2-}), forming methane gas (CH_4). For more detailed information on microbial metabolism the reader is referred to, for example, Grant and Long (1985), Kuenen and Robertson (1987), Laanbroek (1990), and Paul and Clark (1996) (see also Chapters 8, 9, and 11).

Wetland Algae

The assemblage of primitive plants that are collectively referred to as algae includes a tremendously diverse array of organisms. Algae may size from single cells as small as one micrometer to large seaweeds which may grow to over 50 meters. Many of the unicellular forms are motile, and may intergrade confusingly with the Protozoa (South and Whittick, 1987). Algae are ubiquitous; they occur in every kind of water habitat (freshwater, brackish, and marine). However, they can also be found in almost every habitable environment on earth—in soils, permanent ice, snow fields, hot springs, and hot and cold deserts.

Algae may be an important component of a treatment wetland, either as an early colonizing community or as the intended dominant design community. The reader is referred to Vymazal (1995) for a more complete description of algae and element cycling in wetlands.

Algae are unicellular or multicellular, photosynthetic organisms that do not have the variety of tissues and organs of higher plants. Algae are a highly diverse assemblage of species that can live in a wide range of aquatic and wetland habitats. Many species of algae are microscopic and are only discernable as the green or brown color or "slime" occurring on submerged substrates or in the water column of lakes, ponds, and wetlands. Other algal species develop long, intertwined filaments of microscopic cells that look like mats of hair-like seaweed, submerged or floating in ponds and shallow water environments.

For the most part, algae depend on light for their metabolism and growth and serve as the basis for an autochthonous foodchain in aquatic and wetland habitats. Organic compounds created by algal photosynthesis contain stored energy, which is used for respiration or which enters the aquatic foodchain and provides food to a variety of microbes and other heterotrophs. Alternatively, this reduced carbon may be directly deposited as detritus to form organic peat sediments in wetlands and lakes.

Algae also depend on an ample supply of the building blocks of growth including carbon, typically extracted from dissolved carbon dioxide in the water column, and on macro and micronutrients essential to all plant life. When light and nutrients are plentiful, algae can create massive populations and contribute significantly to the overall food web and nutrient cycling of an aquatic or wetland ecosystem. When shaded by the growth of macrophytes, algae frequently play a less important role in wetland energy flows.

Most species of algae need ample water during some or all of their life cycles. Because water quality and climatic variables such as air and water temperature and light intensity are the principal determinants of algal species distribution, the algal flora of wetlands is generally similar to the regional algal flora living in ponds, lakes, springs, streams, rivers, and similar aquatic environments. The algal flora of wetlands differs from the flora of more aquatic environments primarily in response to varying water chemistry, water depth, light inhibition by emergent macrophytes, and seasonal desiccation which is more likely in shallow water environments.

Classification

Algae comprise a very diverse group of organisms that, since the earliest times, defied precise definition. Bold and Wynne (1985) wrote:

> The term "algae" means different things to different people, and even the professional botanist and biologist find algae embarrassingly elusive to define. The reasons for this are that algae share their more obvious characteristics with other plants, while their really unique features are more subtle.

Algae may be classified by evolutionary or genetic relationships, morphological adaptations, or by ecological functions. Taxonomic identification of algae in wetlands rarely is required to design or operate wetland treatment systems. For detailed taxonomy of this phylum, the reader is referred to Lee (1980), South and Whittick (1987), and Vymazal (1995). Two general schemes for classification of aquatic algae (and microorganisms in general) can be found in the literature (Vymazal, 1995).

One scheme is a two-component system, as follows:

- Plankton: organisms that swim or float in the water
- Benthos: organisms that grow on the bottom of the water body

The second and older system makes a distinction within the attached (epiphytic) component:

- Periphyton: all aquatic organisms that grow on submerged substrates
- Benthos: organisms that grow on the bottom of the water body

Other designations include metaphyton, which is the community of floating algae.

Plankton

Reynolds (1984) characterize plankton as the "community" of plants and animals adapted to suspension in the sea or in fresh waters and which is liable to passive movement by wind and current. Planktonic organisms are suspended in the water column and lack the means to maintain their position against the current flow, although many of them are capable of limited, local movement with the water mass. Phytoplankton occur in virtually all bodies of water. All algal groups except the Rhodophyceae, Charophyceae, and Phaeophyceae contribute species to the phytoplankton flora. Phytoplankton encompasses a surprising range of cell size and cell volume from the largest forms visible to the naked eye, (e.g., *Volvox* [500–1500 μm]) in the freshwater and *Coscinodiscus* species in the ocean, to the algae as small as 1 μm in diameter (Vymazal, 1995). Phytoplankton algae are mainly unicellular, though many colonial and filamentous forms occur, especially in fresh waters. Example photographs of wetland phytoplankton algae may be found in Vymazal (1995) and in Fox *et al.* (1981) for domestic wastewater. Planktonic or free-floating algae are generally not important in wetland ecosystems unless open or deep water areas are present. Plankton spend most of their life cycle suspended in the water column and are the most important algal component in lakes and some ponds. Tychoplankton (pseudoplankton) are algae that initially grow as attached species and which subsequently break free from their substrate and live planktonically for part of their life cycle. Tychoplanktonic algal species are most common in streams and in littoral wetlands.

Plankton are probably not important as a component of pollutant processing in most wetlands. However, the use of emergent wetlands to shade out and remove plankton from facultative pond effluents is an important treatment wetland consideration.

Attached Algae

As far as the attached algal communities are concerned, there are three overlapping terms used to describe algae growing attached to any kind of substrates: benthos, periphyton, and aufwuchs. In the literature, there is a lot of confusion and controversy about these terms (Vymazal, 1995). Benthos is composed of attached and bottom-dwelling organisms (Bold and Wynne, 1985). Epiphytic algae grow attached to various substrates and may be classified as:

- Epilithic (growing on stones)
- Epipelic (attached to mud or sand)
- Epiphytic (attached to plants)
- Epizoic (attached to animals)

Periphyton in its broad definition includes all aquatic organisms (microflora) growing on submergent substrates. Although periphyton usually begin colonization of new plant surfaces by attached algal growth of filamentous and unicellular species, this functional component also includes a variety of free-living algae (not attached to the surface), fungi, bacteria, and protozoans following a period of maturation. Periphyton growing on plants is often called epiphyton. Aufwuchs is a more general term than periphyton and includes all algae and associated microscopic life attached to all surfaces in an aquatic or wetland system. These surfaces frequently include living vascular plants as well as dead plants, leaves, branches, trunks, stones, and exposed substrates. Benthic or attached algae are more specific terms that refer only to the algal component of the periphyton or aufwuchs.

Epiphytic algae generally show little substrate specificity; many epiphytic species are encountered in natural epilithic communities and on artificial substrates. In spite of seeming relative indifference of epiphytic algae to their substrate, the epiphytic habitat has several distinctive attributes. The surface itself has a definite life span. New leaves are colonized as they develop during the growing season resulting in a summer and autumn peak in epiphytic biomass and productivity. The canopy of aquatic macrophytes often creates light-limiting conditions for epiphytic algae (Darley, 1982). On the other hand, decreases in growth and photosynthetic rates, as well as abundance and occurrence of submersed macrophytes, have been attributed to light attenuation by the periphyton complex (Vymazal, 1995).

In their use of nutrients from the sediment (via macrophyte tissue) as well as from the overlying water, epiphytes can play an important role in nutrient cycling. Much of the physiological research on epiphytic algae has focused on the question of nutrient transfer from rooted, aquatic, vascular plants to their epiphytes. A few studies have demonstrated a transfer of organic carbon, nitrogen, and phosphorus from macrophyte to the epiphytic community. Experiments with radio-labeled phosphorus show that this release is small for macrophytes in active growth (3–24%), though larger proportions (60%) can apparently be obtained by firmly attached epiphytic algae when phosphorus availability in the water phase is extremely low (Cattaneo and Kalff, 1979; Moeller *et al.*, 1988) The release is probably larger from senescent leaves, but perhaps of little significance because old leaves are subsequently shed (Sand-Jensen *et al.*, 1982). There is evidence that some rooted aquatic plants act as pumps, transferring phosphorus and other nutrients from the sediments to epiphytes and the water column. The amount of nutrient released, however, is very small (Cattaneo and Kalff, 1979).

Interactions between epiphytic algae and their host macrophytes have been subject to controversy. Competing hypotheses differ as to whether (1) the host macrophyte is a neutral substrate or (2) the host macrophyte influences epiphyton production and community composition by mechanisms independent of morphology. Similarities between natural and artificial macrophyte-substrates in community composition, biomass, and production of colonizing epiphyton support the former hypothesis. On the other hand, it has been found that epiphyton species composition and abundance were related to the macrophyte-mediated changes in the physicochemical environment. The responses of epiphytic and epipelic algae to primary physical, chemical, and biotic parameters have been discussed in detail by Wetzel (2001). Photographic examples of attached algae are given in Vymazal (1995).

Filamentous Algae

Filamentous algae that occur in wetlands as periphyton or mats may dominate the overall primary productivity of the wetland, controlling dissolved oxygen and carbon dioxide concentrations within the wetland water column. They are opportunistic, because they can grow very rapidly compared to macrophytes. Therefore, the early period of constructed wetland life may create ideal conditions for algal establishment (Figure 3.1). However, macrophytes can later easily shade out the algae. Diurnal DO profiles in wetlands and other aquatic environments with substantial populations of submerged plants undergo major changes in relation to the daily gross and net productivity. Wetland water column DO can fluctuate from near zero during the early morning following a night of high respiration to well over saturation (>20 mg/L) in high algal growth areas during a sunny day. Dissolved carbon dioxide and consequently the pH of the water vary proportionally to DO because of the corresponding use of CO_2 by plants during photosynthesis and release at night during respiration. As CO_2 is stripped from the water column by algae during the day, pH may rise by 2 to 3 pH

FIGURE 3.1 Algae were the first colonizers of this 25-ha constructed wetland cell near Carson City, Nevada.

units (a 100- to 1,000-fold decrease in H^+ concentration). These daytime pH changes are reversible, and the production of CO_2 at night by algal respiration frequently returns the pH to the previous day's value by early morning.

Algae also store and transform essential growth nutrients in wetlands and aquatic habitats. Because of their relatively low contribution to the overall fixed carbon in wetlands, algae do not constitute a major storage reservoir for these elements in wetlands. However, because of their high turnover rates in some aquatic habitats, algae may be important for short-term nutrient fixation and immobilization with subsequent gradual release and recycling. The functional result of this nutrient cycling is that intermittent high inflow concentrations of pollutants used by algae for growth may be immobilized and transformed more effectively than would be possible without these components, thereby reducing the amplitude of wetland constituent outflow concentrations.

For a detailed description of the importance of algae in wetlands, see Vymazal (1995).

Wetland Macrophytes

Macrophytic plants provide much of the visible structure of wetland treatment systems. There is no doubt that they are essential for the high-quality water treatment performance of most wetland treatment systems. The numerous studies measuring treatment with and without plants have concluded almost invariably that performance is higher when plants are present. This finding led some researchers to conclude that wetland plants were the dominant source of treatment because of their direct uptake and sequestering of pollutants. It is now known that plant uptake is the principal removal mechanism only for some pollutants, and only in lightly loaded systems. During an initial successional period of rapid plant growth, direct pollutant immobilization in wetland plants may be important. For many other pollutants, plant uptake is generally of minor importance compared to microbial and physical transformations that occur within most wetlands. Macrophytic plants are essential in wetland treatment systems because they provide the structure that fosters many removal processes.

The term macrophyte includes vascular plants that have tissues that are easily visible. Vascular plants differ from algae through their internal organization into tissues resulting from specialized cells. A wide variety of macrophytic plants occur naturally in wetland environments. The United States Fish and Wildlife Service has more than 6,700 plant species on their list of obligate and facultative wetland plant species in the United States. Godfrey and Wooten (1979; 1981) list more than 1,900 species (739 monocots and 1,162 dicots) of wetland macrophytes in their taxonomy of the southeastern United States. Obligate wetland plant species are defined as those which are found exclusively in wetland habitats, whereas facultative species are those that may be found in upland or in wetland areas. There are many guidebooks that illustrate wetland plants (for example, Hotchkiss, 1972; Niering, 1985; Cook, 1996). Lists of plant species that occur in wetlands are available (e.g., RMG, 1992).

Wetland macrophytes are the dominant structural component of most wetland treatment systems. A basic understanding of the growth requirements and characteristics of these wetland plants is essential for successful treatment wetland design and operation.

Classification

The plant kingdom is divided taxonomically into phyla, classes, and families, with certain families either better represented or occurring only in wetland habitats. The major plant phyla are the mosses and clubmosses (*Bryophyta*) and the vascular plants (*Tracheophyta*). In the vascular plant phylum there are three important classes of plants: ferns (*Filicinae*), conifers (*Gymnospermae*), and flowering plants (Angiospermae). The flowering plants are further divided into the monocots (*Monocotyledonae*) and dicots (*Dicotyledonae*).

Because plant taxonomic families were developed to provide insight into the evolutionary affinity of plant species, it is not surprising that some families are well represented by multiple obligate wetland species. Vascular plants including wetland plants may also be categorized morphologically by descriptors such as woody, herbaceous, annual, or perennial. Woody species have stems or branches that do not contain chlorophyll. Because these tissues are adapted to survive for more than one year, they are typically more durable or woody in texture. Herbaceous species have aboveground tissues that are leafy and filled with chlorophyll-bearing cells that typically survive for only one growing season. Woody species include shrubs that attain heights up to 2 or 3 m and trees that generally are more than 3 m in height when mature.

Annual plant species survive for only one growing season and must be reestablished annually from seed. Perennial plant species live for more than one year and typically propagate each year from perennial root systems or from perennial aboveground stems and branches. Nearly all woody plant species are perennial, but herbaceous species may be annual or perennial.

Four groups of aquatic macrophytes (Figure 3.2) can be distinguished on a basis of morphology and physiology (Wetzel, 2001):

1. Emergent macrophytes grow on water-saturated or submersed soils from where the water table is about 0.5 m below the soil surface to where the sediment is covered with approximately 1.5 m of water (e.g., *Acorus calamus, Carex rostrata, Phragmites australis, Schoenoplectus (Scirpus) lacustris, Typha latifolia*).
2. Floating-leaved macrophytes are rooted in submersed sediments in water depths of approximately 0.5 to 3 m and possess either floating or slightly aerial leaves (e.g., *Nymphaea odorata, Nuphar luteum*).
3. Submersed macrophytes occur at all depths within the photic zone. Vascular angiosperms (e.g., *Myriophyllum spicatum, Ceratophyllum demersum*) occur only to about 10 m (1 atm hydrostatic pressure) of water depth and nonvascular macroalgae occur to the lower limit of the photic zone (up to 200 m, e.g., *Rhodophyceae*).
4. Freely floating macrophytes are not rooted to the substratum; they float freely on or in the water and are usually restricted to nonturbulent, protected areas (e.g., *Lemna minor, Spirodella polyrhiza, Eichhornia crassipes*).

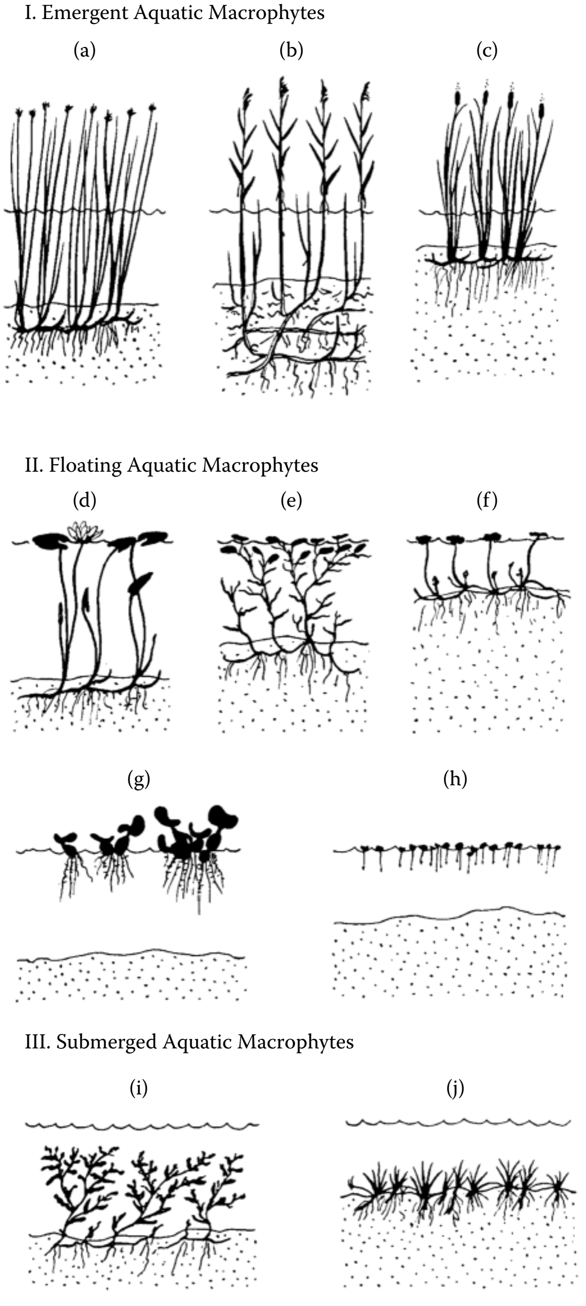


FIGURE 3.2 Sketch showing the dominant life forms of aquatic macrophytes. The species illustrated are (a) *Scirpus* (*Schoenoplectus*) *lacustris*, (b) *Phragmites australis*, (c) *Typha latifolia*, (d) *Nymphaea alba*, (e) *Potamogeton gramineus*, (f) *Hydrocotyle vulgaris*, (g) *Eichhornia crassipes*, (h) *Lemna minor*, (i) *Potamogeton crispus*, (j) *Littorella uniflora*. (From Brix and Schierup (1989b). *Ambio* 18: 100–107. Reprinted with permission.)

In addition, a large number of the emergent macrophytes can be established in floating mats, either with or without a supporting structure. Some species have one or more of these growth forms; however, there is usually a dominant form that enables the plant species to be classified. In emergent plant species, most of the aboveground part of the plant emerges above the water line and into the air.

Both floating and submerged vascular plant species may also occur in wetland treatment systems. Floating species have leaves and stems buoyant enough to float on the water surface. Submerged species have buoyant stems and leaves that fill the niche between the sediment surface and the top of the water column. Floating and submerged species prefer deep aquatic habitats, but they may occur in wetlands when water depth exceeds the tolerance range for rooted, emergent species.

TABLE 3.1
Number of Plant Species by Group Found in Constructed Wetlands in the North American Database, Version 2.0*

Plant Group	Number of Species Recorded
Emergent macrophyte	501
Floating aquatic plant	31
Submerged aquatic plant	10
Shrub	17
Tree	25
Unknown	5
Vine	5
Totals	**594**

* *This database is dominated by FWS wetlands, and covers only a subset of existing systems.*

Source: Data from NADB database (1998) *North American Treatment Wetland Database (NADB), Version 2.0.* Compiled by CH2M Hill, Gainesville, Florida.

TABLE 3.2
Dominant Plant Species Found in Constructed Treatment Wetlands

Common Name	Scientific Name
Bacopa	*Bacopa caroliniana*
Bulrush	*Scirpus* spp.
Cattail	*Typha* spp.
Common reed	*Phragmites australis*
Coontail	*Ceratophyllum demersum*
Duck potato	*Sagittaria* spp.
Duckweed	*Lemna* spp.
Frogs-bit	*Limnobium spongea*
Pennywort	*Hydrocotyle* spp.
Pickerelweed	*Potederia* spp.
Pondweed	*Potamogeton* spp.
Reed canary grass	*Phalaris arundinacea*
Softrush	*Juncus* spp.
Spatterdock	*Nuphar luteum*
Water hyacinth	*Eichhornia crassipes*
Waterweed	*Elodea* spp.

Source: Modified from NADB database (1998) *North American Treatment Wetland Database (NADB), Version 2.0.* Compiled by CH2M Hill, Gainesville, Florida.

Table 3.1 lists the classes of plants reported in treatment wetlands and their numbers. Table 3.2 lists the dominant plants in treatment wetlands.

Adaptations to Life in Flooded Conditions

Prolonged flooding or waterlogging restricts oxygen movement from the atmosphere to the soil. Diffusion can occur but it is 10,000 times slower in saturated soils than it is in aerated soils (Greenwood, 1961). Upon flooding, respiration by aerobic bacteria and other organisms consume the oxygen remaining in the soil within hours to days (Pezeshki, 1994). Soil oxygen deficiency (partial hypoxia, complete anoxia) poses the main ecological problem for plant growth as it affects plant functions such as stomatal opening, photosynthesis, water and mineral uptake, and hormonal balance (Kozlowski, 1984b). Life in permanently or periodically anaerobic soils or substrates is more difficult than living in mesic soils due to the nature of a highly reduced environment (low redox potential), possibly together with soluble phytotoxins (Tiner, 1999).

A wide range of adaptations make it possible for plants to grow in water or wetlands. These adaptations include physiological responses, morphological adaptations, behavioral responses, reproductive strategies, and others (Table 3.3). Major plant adaptations in free water surface (FWS) and subsurface constructed wetlands are shown in Figures 3.3 and 3.4. For a detailed description of macrophyte adaptations and responses to flooding see Hook and Crawford (1978), Kozlowski (1984a), Crawford (1987), Hejný and Hroudová (1987), or Jackson *et al.* (1990).

One of the most important adaptations to flooding is the development of aerenchymous plant tissues (Figure 3.5) that transport gases to and from the roots through the vascular tissues of the plant above water and in contact with the atmosphere, providing an aerated root zone and thus lowering the plant's reliance on external oxygen diffusion through water and soil (Armstrong, 1978; Jackson and Drew, 1984; Zimmerman, 1988; Brix, 1993). Lenticels or small openings on the above water portions of these plants provide an entry point for atmospheric oxygen into this aerenchymous tissue network. Lenticel surface area may be increased through plant growth, height increases, or the formation of swollen buttresses in trees and woody herbs and in cypress knees.

Plant survival in flooded environments is a balance between the severity of oxygen limitation and the adaptations available to overcome this oxygen shortage. Thus, hydrophytic plants may be adapted to survive and even grow in specific flooded conditions, such as three months each year, or in "clean" or flowing water, which might have higher in situ dissolved oxygen concentrations (Gosselink and Turner, 1978). However, these same plants may not be able to grow or survive during five months of flooding or in stagnant or "dirty" water conditions. This is shown in Figure 3.3. Likewise, plants may have adaptations that allow prolonged survival in one foot of water but not at two feet. It may be hypothesized that this balance is tilted unfavorably at higher water levels because of reduced aerial plant stem surface area to provide oxygen to the roots

TABLE 3.3
Plant Adaptations or Responses to Flooding and Waterlogging

Morphological	Stem hypertrophy (e.g., buttressed tree trunks); large air-filled cavities
Adaptations/responses	In the center (stele) of roots and stems; aerenchyma tissue in roots and other plant parts; hollow stems; shallow root systems; adventitious roots; pneumatophores (e.g., cypress knees); swollen, loosely packed root nodules; lignification and suberization (thickening) of roots; soil water roots; succulent roots; aerial root-tips; hypertrophied (enlarged) lenticle; relatively pervious cambium (in woody species); heterophylly (e.g., submerged versus emergent leaves on same plants); succulent leaves.
Physiological adaptations	Transport of oxygen to roots from lenticles and/or leaves (as often evidenced by oxidized rhizospheres); anaerobic respiration; increased ethylene production; reduction of nitrate to nitrous oxide and nitrogen gas; malate production and accumulation; reoxidation of NADH; metabolic adaptations
Other adaptations/responses	Seed germination under water; viviparous seeds; root regeneration responses (e.g., adventitious roots); growth dormancy (during flooding); elongation of stem or petioles; root elongation; additional cell wall structures in epidermis or cortex; root mycorhizzae near upper soil surface; expansion of coleoptiles (in grasses); change in direction of root or stem growth (horizontal or upward); long lived seeds; breaking of dormancy of stem buds (may produce multiple stems or trunks).

Source: From Tiner (1999) *A Guide to Wetland Identification, Delineation, Classification, and Mapping.* CRC Press, Boca Raton, Florida.

through the lenticels and aerenchymous tissues. This proposed explanation is supported by the finding that hydrophytes generally respond to flooding by growing taller, a growth response that allows a more favorable balance between emergent and submerged plant organs (Grace, 1989).

Hydropattern

The term hydropattern refers to the time series of water depths in the wetland. The concept of hydropattern, or water regime, includes two interdependent components: (1) the duration of flooded or saturated soil conditions (the hydroperiod

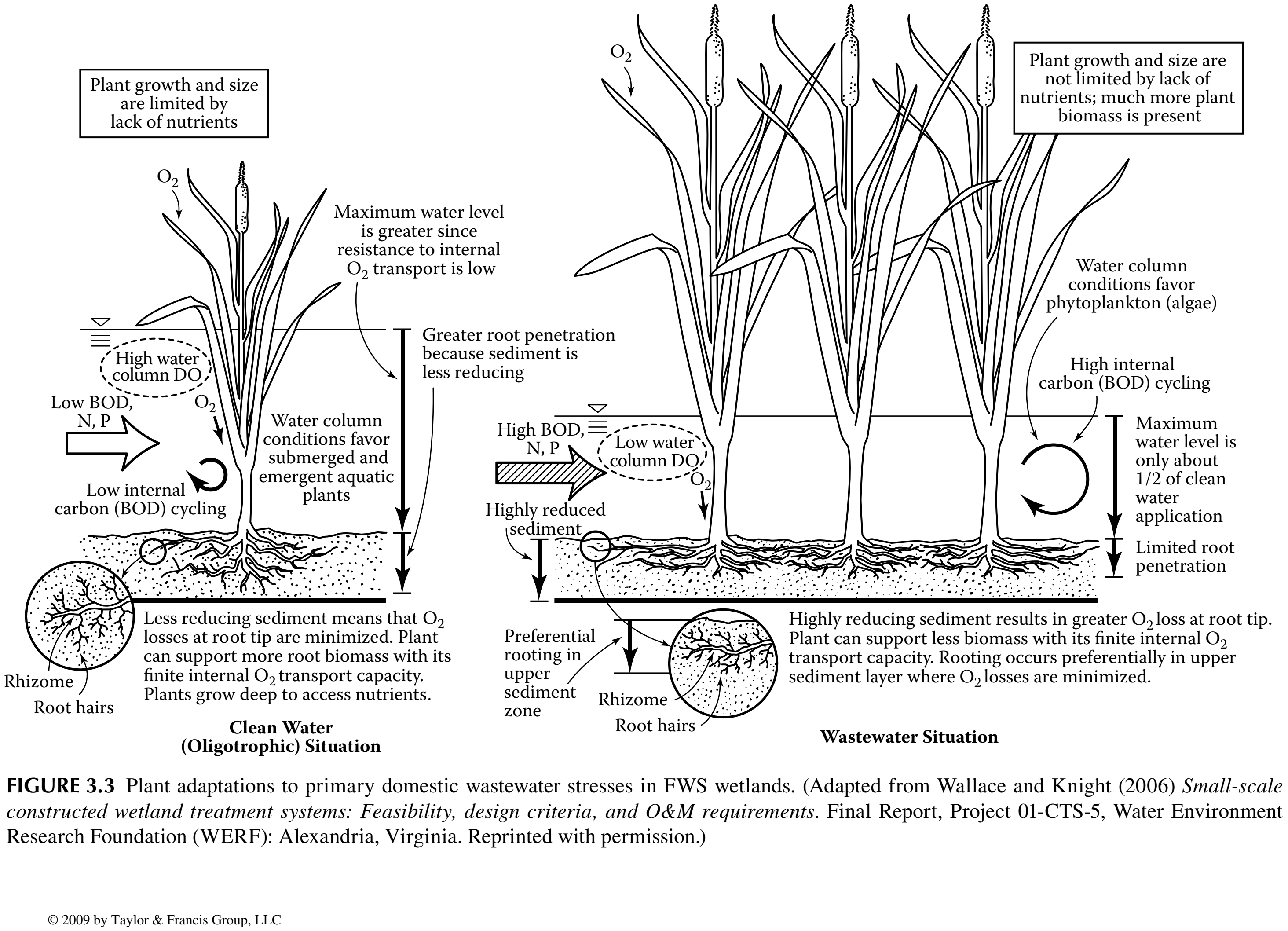


FIGURE 3.3 Plant adaptations to primary domestic wastewater stresses in FWS wetlands. (Adapted from Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

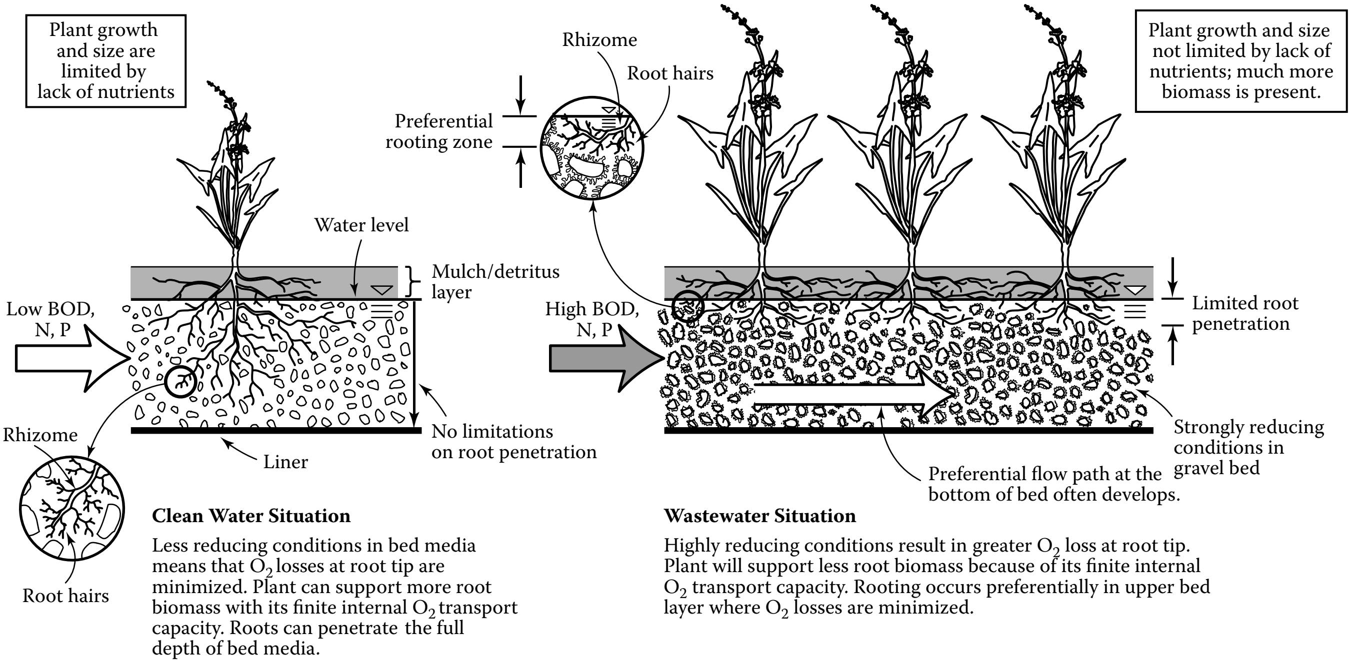


FIGURE 3.4 Plant adaptations to primary domestic wastewater stresses in HSSF wetlands. (Adapted from Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

as a percentage of time with flooding), and (2) the depth of flooding (Gunderson, 1989). Although hydroperiod refers to the duration of flooding, the term water regime refers to hydroperiod as well as to the combination of water depth and flooding duration (depth-duration curve). The duration and depth of flooding affect plant physiology because of soil oxygen concentration, soil pH, dissolved and chelated macro and micronutrients, and toxic chemical concentrations. Figure 3.6 uses a graph of water level within a wetland over an annual period to illustrate these two aspects of hydroperiod and water regime. Duration of flooding refers to the percentage of time that a wetland site is flooded or saturated, and depth of flooding refers to the minimum, average, and maximum depths of water at a given or typical spot within

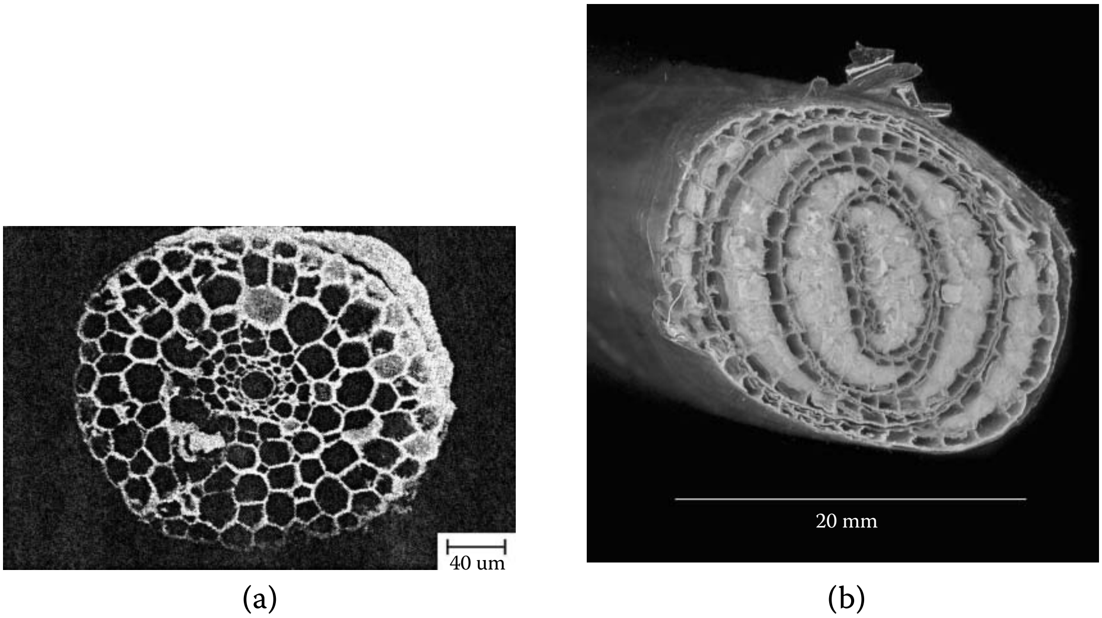


FIGURE 3.5 (a) Internal gas passages in a *Phragmites* root. (From Armstrong and Armstrong (1990b) In *Constructed Wetlands in Water Pollution Control*. Cooper and Findlater (Eds.), Pergamon Press, Oxford, United Kingdom, pp. 529–534. Reprinted with permission.) (b) Internal gas passages in a *Typha* culm.

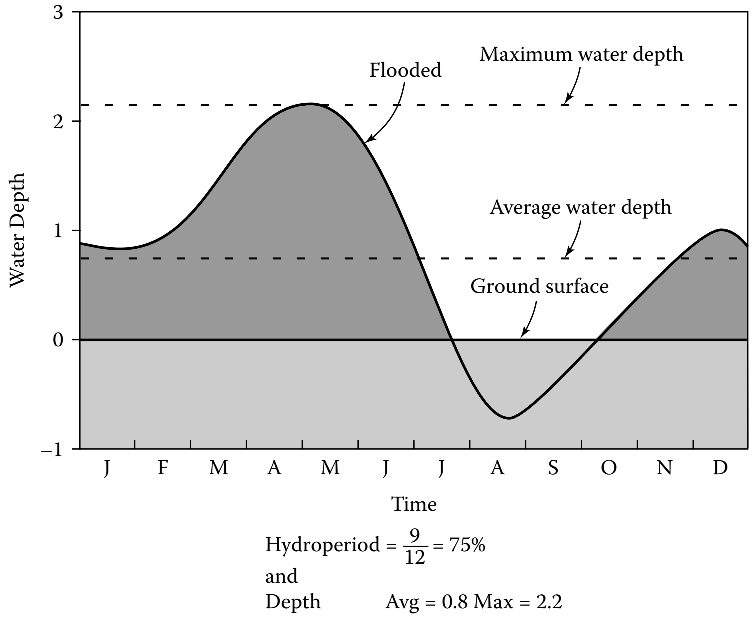


FIGURE 3.6 Components of hydropattern: hydroperiod and wetland water regime. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

a wetland. Hydroperiod curves provide a convenient method for estimating the percentage of time that a wetland is flooded at any water depth and can summarize water level data over any period of record. Note that water level charts and depth-duration curves also can summarize the time and depth that water is located below the ground surface.

Although the presence of water separates uplands from wetlands and aquatic ecosystems, hydropattern is the most important contributor to wetland type or class (Gosselink and Turner, 1978; Gunderson, 1989). The importance of this factor in wetland treatment system design and operation cannot be overstated because incorrect understanding of the hydroperiod and water regime limitations of wetland plant species is a frequent cause of vegetation problems in natural and constructed wetlands. Measuring the hydroperiod is relatively easy. However, selecting the optimal hydroperiod for wetland treatment design and performance is complex.

Oxygen Transport as a Treatment Function

In order to survive in the saturated rooting environment, emergent wetland plants transport oxygen from their leaves down through their stalks to the root tissue (Armstrong, 1979). Because the aerenchyma passageways have occasional blockages to prevent flooding if the root tissues are damaged, internal transport of oxygen is a diffusion-limited process. Some plant species can increase oxygen transport by convective flow of gases (Brix, 1990; Armstrong and Armstrong, 1990a; Brix, 1994b). Dead and broken shoots and stubble also form air pipes to the root zone. Of interest here is the fact that significant quantities of oxygen pass down through the airways to the roots (Brix and Schierup, 1990; Brix, 1993); and that significant quantities of other gasses, such as carbon dioxide and methane, pass upward from the root zone. Internal gas spaces in a *Phragmites* root and a *Typha* culm are shown in Figure 3.5.

The oxygen is used for root respiration and to help detoxify the environment encountered by the growing root tip. Consequently, there are limits as to how far plants can propagate their root systems in a highly reducing environment (Armstrong *et al*., 1990). Some—probably most—of the oxygen passing down the plant into the root zone is used in plant respiration (Brix, 1990). The excess supply of O_2 over that required for plant respiration is termed the *plant aeration flux* (PAF), has been the subject of many research endeavors (Armstrong *et al*., 1990; Brix, 1990; Gries *et al*., 1990; Sorrell *et al*., 2000; Wu *et al*., 2001; Bezbaruah and Zhang, 2003). The difficulty of measuring processes and concentrations in the root microzones has been a major factor in the widely disparate estimates of PAF (Kadlec and Knight, 1996).

Chemical conditions in the root zone are important determinants of the potential for significant PAF (Sorrell, 1999). Hydroponic studies most often create root environments that do not include a significant sediment oxygen demand. Roots are numerous under such conditions, and exchange oxygen along much of their length (Armstrong *et al*., 1990). The morphology and physiology of roots is very different in the anaerobic environment often associated with treatment wetland soils. Under treatment conditions, the number of roots is significantly less than in clean soil or hydroponic conditions. Roots become armored along much of their length, and O_2 losses to the soil and water occur only in a small apical region (Brix, 1994c).

Oxygen transfer by plants was initially thought to be a dominant mechanism in SSF wetland treatment (Kickuth and Könemann, 1987), but recent work has demonstrated that the vast majority of the oxygen transferred by the plant is used for root metabolism, and the amount released to the rhizosphere is small. Different test methods yield different results, but a value of 0.02 $g/m^2 \cdot d$ has been established in two independent studies (Brix and Schierup, 1990; Wu *et al*., 2001). As a result, most modern designers have abandoned the concept of plants acting as "solar powered aerators." Since studies have proven plant-induced oxygen transfer rates to be so small, current design guidelines recommend assuming that oxygen delivered to the wastewater by the plant roots is negligible (U.S. EPA, 2000a). For a further discussion of root aeration, see Chapter 5.

3.2 BIOMASS AND GROWTH

The term *biomass* is most frequently defined as the mass of all living tissue at a given time in a given unit of Earth's surface (Lieth and Whittaker, 1975). It is commonly divided into belowground (roots, rhizomes, tubers, etc.) and aboveground biomass (all vegetative and reproductive parts above the ground level). The term *standing crop* includes live parts and dead parts of live plants that are still attached. These dead parts of plants together with still standing dead plants are called *standing dead*. The term *litter* refers to those dead parts of the plant that have fallen on the ground or sediment, but in some cases also includes standing dead. These compartments exchange material, but not uniformly, over the course of the year (Figure 3.7).

Peak standing crop is defined as the single largest value of plant material present during a year's growth (Richardson and Vymazal, 2001). In tropical communities, with an almost constant biomass, it is not profitable to search for an annual maximum (Westlake, 1969). However, in all other climatic regions the biomass fluctuates widely throughout the year (Dykyjová and Kvet, 1978; Shew *et al*., 1981; Kaswadji *et al*., 1990). The range of standing crop of wetland plants is quite large (Kvet, 1982; Mitsch and Gosselink, 1993; Vymazal, 1995). Another terminology has been advanced by Mueleman *et al*. (2002), which suggests that the total is *phytomass*, which is composed of living material (biomass) and dead (necromass).

Gross Primary Production (also called *Gross Primary Productivity*, or GPP) is normally defined as the assimilation of organic matter by a plant community during a specified period, including the amount used by plant respiration. *Net Primary Production*, or NPP, is defined as the biomass that is incorporated into a plant community during a specified time interval, less that respired. This is the quantity that is measured by harvest methods and which has also been called *net assimilation* or *apparent photosynthesis*. The term *Net Aerial (or Aboveground) Primary Production* (NAPP) is defined as the biomass incorporated into the aerial parts (leaf, stem, seed, and associated organs) of the plant community (Milner and Hughes, 1968).

NPP of freshwater marshes is estimated most frequently through harvest of annual peak standing stocks of live and dead plant biomass. When root biomass is measured, it is usually an important part of net annual plant production. Some researchers consider net primary productivity estimates by peak standing stock to be underestimates because they do not account for biomass turnover during the growing season (Pickett *et al*., 1989). Kvet (1982) estimates turnover rates (productivity/biomass) in the range of 1.1–1.5 for submerged species, 1.05–1.5 for short emergent species, 1.05–1.3 for tall emergent species, and 1.15 for tall graminoids. For comparison, phytoplankton has a turnover rate in the range of 450–600. Table 3.4 summarizes some typical estimated net production data from wetland ecosystems, both natural and treatment.

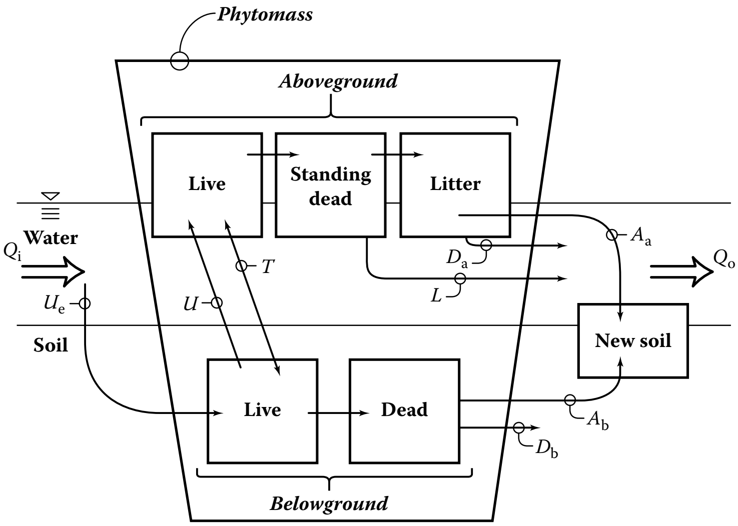


FIGURE 3.7 Transfers of materials in the biosphere of wetlands. Biomass consists of living, above and below ground components. Necromass consists of dead roots and rhizomes, plus aboveground standing dead and litter. Phytomass is the combination of biomass and necromass. Transfer to the phytomass occurs by external plant uptake (U_e). Transfer back to surface water and porewater occurs via leaching (L) and decomposition (D_a and D_b). Necromass residuals lose their identity, and accrete as new soils and sediments (A_a and A_b).

TABLE 3.4
End of Season Plant Biomass in Wetlands

Species	Location	Reference	Water	S/P/E	Live Above (g/m^2)	Total Above (g/m^2)	Roots and Rhizomes (g/m^2)
Cattails							
Typha latifolia	Wisconsin	Smith *et al.* (1988)	N	105/245/290	—	1,400	450
Typha latifolia	Texas	Hill (1987)	N	60/240/345	—	2,500	2,200
Typha glauca	Iowa	van der Valk and Davis (1978)	N	120/265/290	2,000	—	1,340
Typha latifolia	Michigan	Unpublished data from Houghton Lake	N	120/245/275	490	890	6,200
Typha latifolia	Michigan	Unpublished data from Houghton Lake	S	120/245/275	1,240	2,310	2,900
Typha angustifolia	Michigan	Unpublished data from Houghton Lake	S	120/245/275	1,886	3,615	—
Typha latifolia	Kentucky	Pullin and Hammer (1989)	P	—	5,602	—	3,817
Typha angustifolia	Kentucky	Pullin and Hammer (1989)	P	—	5,538	—	4,860
Bulrushes							
Scirpus fluviatilis	Iowa	van der Valk and Davis (1978)	N	130/265/285	790	—	1,370
*Scirpus validus**	Iowa	van der Valk and Davis (1978)	N	120/210/300	2,100	—	1,520
Scirpus validus	New Zealand	Tanner (2001a)	P	30/205/350	2,100	2,650	1,200
Scirpus validus	Kentucky	Pullin and Hammer (1989)	P	—	—	2,355	7,376
Scirpus cyperinus	Kentucky	Pullin and Hammer (1989)	P	—	—	3,247	12,495
Phragmites							
Phragmites australis	U.K.	Mason and Bryant (1975)	N	75/220/305	942	1,275	—
Phragmites australis	Iowa	van der Valk and Davis (1978)	N	—	—	1,110	1,260
Phragmites australis	Netherlands	Mueleman *et al.* (2002)	N	105/255/350	2,900	3,200	7,150
Phragmites australis	Brisbane	Greenway (2002)	S	—	1,460	2,520	1,180
Phragmites australis	Netherlands	Mueleman *et al.* (2002)	P	105/255/355	5,000	5,500	3,890
Phragmites australis	New York	Peverly *et al.* (1993)	L	100/270/330	10,800	—	8,700

Note: Water type is N = no wastewater, S = nutrients at secondary treatment levels, P = nutrients at primary treatment levels, L = landfill leachate at around 300 gN/m^3. S/P/E refers to the start, peak, and end yeardays of the growing season (add 182 days for Southern Hemisphere).

*Currently known as *Schoenoplectus tabernaemontani*.

Primary productivity of wetland plants is increased by the availability of water, light, and nutrients. Adding wastewater to wetlands generally increases the availability of water and nutrients and consequently results in the stimulation of gross and net primary productivity of these ecosystems (Guntenspergen and Stearns, 1981; Nixon and Lee, 1986).

Fertilizer Response

The growth of wetland plants, like that of terrestrial plants, is stimulated by fertilization (Boyd, 1971; Jordan *et al*., 1999; Mueleman *et al*., 2002). When a wetland becomes the recipient of waters with higher nutrient content than those it has been experiencing, there is a response of the vegetation, both in species composition and in total biomass. This response has been detailed for the Houghton Lake wetland by Kadlec and Alvord (1989). The increased availability of nutrients produces more vegetation during the growing season, which in turn means more litter during the nongrowing season. This litter requires several years to decay, and hence the total pool of living and dead material grows slowly over several years to a new and higher value. A significant quantity of structural components are thus retained in the wetland.

Primary productivity of wetland plants is increased by the availability of water, light, and nutrients. Adding wastewater to wetlands generally increases the availability of water and nutrients and consequently results in the stimulation of gross and net primary productivity of these ecosystems. Figure 3.8 illustrates the typical plant growth response curve to increased concentrations of nitrogen and phosphorus. The maximum rate of plant growth is attained as nutrient levels are initially increased. However, at higher nutrient levels, plant growth levels off while luxury nutrient uptake continues, and at higher nutrient concentrations, phytotoxic responses are observed.

Figure 3.9 gives an example of this fertilizer response for soft-stemmed bulrush, *Schoenoplectus (Scirpus) validus*, grown in dairy wastewater. As the nitrogen concentration was increased, both above- and belowground biomass increases (Tanner, 1994). However, there is a suggestion of a maximum

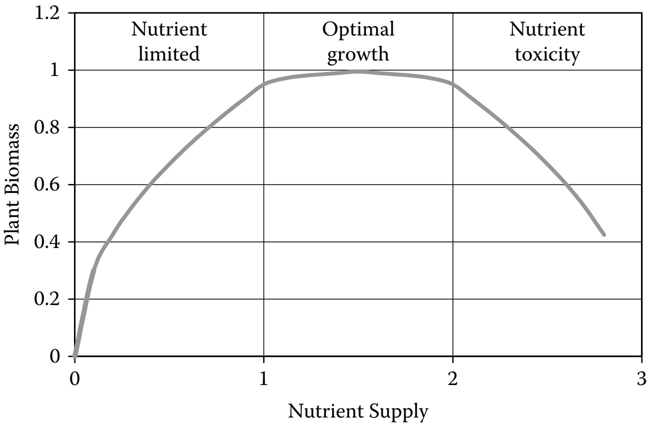


FIGURE 3.8 General relationship between plant biomass and nutrient concentration in the water column.

at the highest concentrations. In fact, root death was noted by Tanner (1994) in plants growing in piggery wastewaters, where high ammonia concentrations (mean 222 mg/L) were at potentially phytotoxic levels. For example, ammonia concentrations of 200 mg/L are known to be detrimental to water hyacinths (de Casabianca-Chassany *et al.*, 1992). Other studies have also established similar effects for other treatment wetland plants. Hill *et al.* (1997) found dry matter production of *Typha latifolia, Phragmites australis,* and *Sagittaria latifolia* were unaffected by ammonia in the concentration range 20–80 mg/L range. Dry matter production of *Schoenoplectus* (*Scirpus*) *acutus* was found to be maximized in the 30–50 mg/L range, and then to rapidly fall off above 60 mg/L.

Seasonal Patterns

The growth and senescence of the soft tissue macrophytes commonly used for wastewater treatment all follow a common seasonal pattern in temperate climates. In northern climates, growth begins at the time of frost disappearance (around April), and senescence begins in early autumn (around September). This autumnal decline creates standing dead aboveground plant material, which subsequently in part decomposes, and in part falls to the soil surface.

A specific case for *Typha* is shown in Figure 3.10, which is representative of other emergent macrophytes as well. New growth proceeds from small shoots that may be initiated as early as late summer of the preceding year for *Typha* (Bernard, 1999), but remain tiny and dormant over the winter season. Aboveground biomass increases rapidly in early spring, typically commencing from late February to Late April, depending on climate. Growth tapers off, causing aboveground biomass to peak in late summer to early autumn. The size of the peak standing crop varies considerably with plant species and degree of nutrient availability (see Table 3.4). Typically, there is some degree of senescence that accompanies the later portions of the growth period, so that the total peak standing crop exceeds the live peak standing crop. During autumn, more rapid senescence occurs, leaving only a residual of standing and/or prostrate aboveground dead material.

Belowground biomass follows a much more muted annual cycle. In some cases, available methods of root and rhizome biomass measurement are not accurate enough to clearly define a pattern (Figure 3.10). In other cases, a midsummer depression has been found, to about 50% of the midwinter maximum (Smith *et al.*, 1988; Mueleman *et al.*, 2002). But mid summer maxima were found for *Sparganium* and *Phragmites* in Iowa (van der Valk and Davis, 1978). When root biomass is measured, it is usually an important part of net annual plant production. NPP estimates by peak standing stock are underestimates because they do not account for biomass turnover during the growing season. For instance, a multiplier of 1.2–1.4 for aboveground cattails and *Spartina* has been reported by Cronk and Fennessy (2001).

In tropical or subtropical climates, seasonality is much more muted (Figure 3.11). There may be periods of dormancy

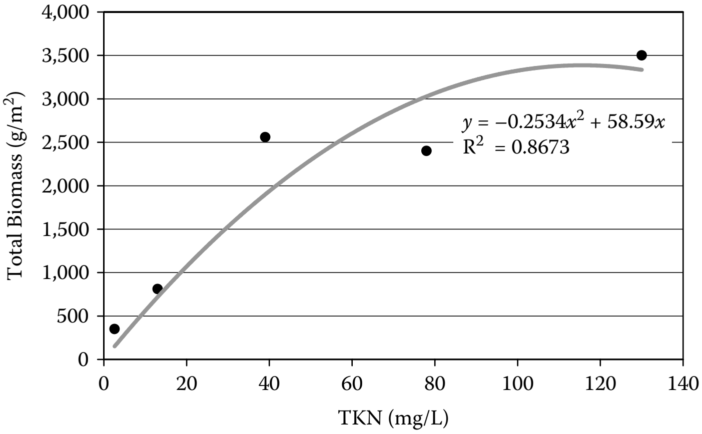


FIGURE 3.9 Growth of *Schoenoplectus* (*Scirpus*) *validus* in dairy wastewater at various dilutions. The accompanying range of total phosphorus concentrations was 0.3–14.8 mg/L. (Data from Tanner (1994) *Aquatic Botany* 47(2): 131–153.)

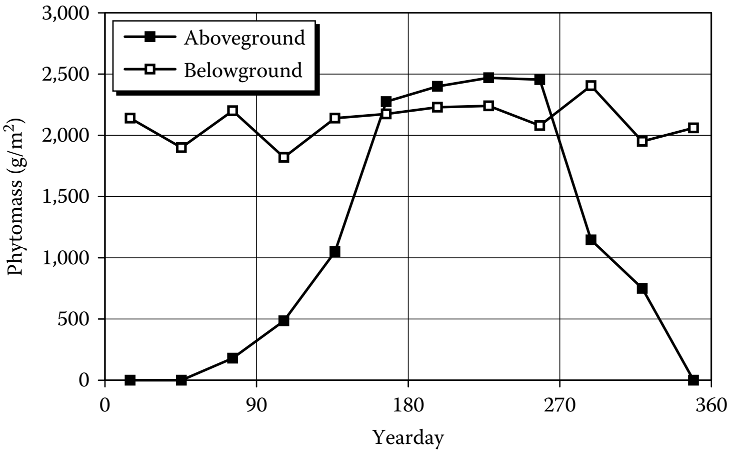


FIGURE 3.10 Seasonal patterns of above- and belowground *Typha angustifolia* phytomass at Richardson, Texas. The climate is warm temperate. Points are averages for two years. (Data from Hill (1987). *Aquatic Botany* 27: 387–394.)

and of regrowth, but there is typically not complete senescence and death of all aboveground plant parts.

Two other factors are important in assessing the growth of wetland plants: the length of the growing season, and belowground productivity. All of the growth for the year occurs in about 100 days in high latitudes, whereas systems in the tropics grow year-round (see Table 3.4). Therefore, the instantaneous growing season rate is much higher than the annualized rate for northern systems. Belowground biomass is typically comparable to aboveground biomass, although the root-to-shoot ratio is sensitive to nutrient status and other variables. The ratio of below to aboveground biomass is generally less in a fertilized environment than in a lower nutrient (natural) environment (Mueleman *et al.*, 2002). Kadlec and Alvord (1989) indicated that belowground biomass responded to fertilization differently from aboveground biomass. The initial vegetation showed greatly reduced root biomass in response to the added nutrients: 1,500 g/m^2 versus 4,000 g/m^2 at the end of the growing season. There are some reports that root growth and activity continues much longer than for aboveground plant parts (Prentki *et al.*, 1978).

Roots and rhizomes persist over winter in northern climates, and therefore standing crop alone is not a measure of productivity. Estimates of turnover times are on the order of two to three years for herbaceous wetland plants. For example, Tanner (2001a) estimated a lifetime of 18 to 24 months for *Schoenoplectus* rhizomes, and Prentki *et al.* (1978) reported 1.5–2 years for *Typha* rhizomes and at least three years for *Phragmites* rhizomes. Therefore, the total growth rate for wetland plants is much higher than for aboveground parts alone.

These factors lead to the conclusion that plant growth is much higher than one standing (aboveground) crop per year. Table 3.5 presents a hypothetical illustration of factors for two climate zones. The growth of plant biomass during the

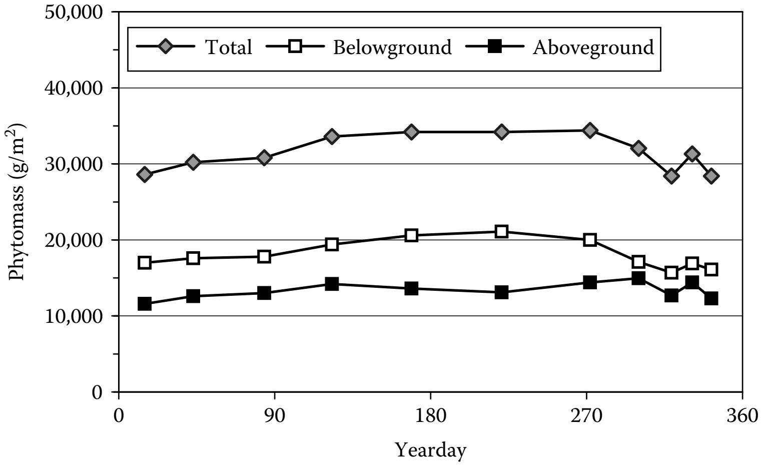


FIGURE 3.11 The pattern of growth of *Phragmites australis* in the warm dry continental climate of Griffith, Australia. The water phosphorus concentration was 12 mg/L, and the approximate annual temperature range was from 10–23°C. (Data from Hocking (1989a) *Australian Journal of Marine and Freshwater Research* 40: 421–444.)

TABLE 3.5
Hypothetical Growth Characteristics of Wetlands Growing in Temperate and Subtropical Conditions

Characteristic	Unit	Temperate Growing Season (M–J–J–A)	Annual	Subtropical Annual
Peak standing crop aboveground	g/m^2	2,000	2,000	2,000
Growth (GPP/NPP = 1.3)	g/m^2	2,600	2,600	—
Growth (4 turnovers per year)	g/m^2	—	—	8,000
Growing season	days	120	365	365
Growth rate above	$g/m^2 \cdot d$	21.7	7.1	21.9
Belowground crop (root/shoot = 1.0)	g/m^2	2,000	2,000	2,000
Growth (0.5 turnovers per year)	g/m^2	1,000	1,000	1,000
Growing season	days	240	365	365
Growth rate below	$g/m^2 \cdot d$	4.2	2.7	2.7
Total growth rate	$g/m^2 \cdot d$	25.8	9.9	24.7
Annualized instantaneous growth rate	$g/m^2 \cdot yr$	9,429	3,600	9,000
Undepleted solar radiation	$MJ/m^2 \cdot d$	38	24	31

Note: These both grow at about the same rate during their respective growing seasons, which are year round in the warm climate.

respective growing seasons is about the same, but the growing season is much attenuated in northern climates. As a result, the annual growth is higher in the warmer environment.

Start-Up: Wetland Vegetation Changes

A constructed wetland begins its existence with the vegetation placed by the constructors, and the seed bank associated with the selected soils. A natural wetland will have evolved over time to contain a mix of vegetation commensurate with the hydropattern and water quality conditions prior to wastewater addition. In either case, the wetland vegetation is likely to change over the course of time, as local adaptations to the treatment hydropattern and quality occur. The plant community that develops over time is a function of organic loading, hydrology, and climate. FWS wetlands that are heavily loaded with organic matter and nutrients will typically develop a less diverse plant community since fewer plant species are able to tolerate the reducing conditions that develop under these circumstances. In polishing wetlands with very high water quality, a diverse species composition may develop.

Individual Plants

Plants reproduce in a two principal ways, by seeding and by vegetative reproduction. A plant starting from seed is a new individual, whereas it is not so easy to identify new individuals when new shoots arise from underground runners. Bulrushes tend to spread in a radial habit, with clumps growing in diameter. Cattails and *Phragmites* tend to spread in a linear mode, with new shoots emerging from a runner at intervals (Figure 3.12). Such runners can extend several meters in just one growing season, for both cattails and *Phragmites*.

Aboveground parts of plants in cold environments have a life span dictated by the photoperiod and frost conditions of the region. They live through one growing season, and new plants emerge the next year, from root stock or from seed. However, in warm climates, individual plants may persist for more than one year. Davis (1989) tagged individual leaves of 43 individual shoots of *Typha domingensis*, and followed their growth over their entire life history in a Florida wetland (Figure 3.13). He found that leaf growth and mortality continued throughout the life span of each tagged plant. New leaves emerged and grew, even while total biomass declined. Older leaves senesced, broke, or died even while total biomass increased. This continual growth and mortality resulted in an annual turnover of 4.4 ± 0.7 times the mean standing crop (Davis, 1989).

The concept of individual plant life history becomes important when, as is a common case, an entire wetland is planted at one time, creating a cohort of plants that will all live about the same length of time. Clearly, without regeneration this wetland will be devoid of plants after a few years. Therefore, the key to a self-sustaining wetland plant community is not only the survivability of plants in the treatment environment, but also the ability to regenerate.

Plant Coverage

The vegetative cover of a treatment wetland refers to the area of wetland plants, and is concerned with four principal measures: (1) fraction areal coverage, (2) stem density, (3) submerged area, and (4) underwater porosity.

Fractional Coverage

Most FWS constructed treatment wetlands are not monotypic communities, but rather contain a patchwork of open water, SAV, EAV, and FAV. In contrast, many SSF systems are in fact completely vegetated with uniform stands of EAV.

FIGURE 3.12 *Phragmites* spreads vegetatively via linear runners. Dr. Hans Brix holds a specimen of only a few weeks' age, in a sludge drying reed bed in Denmark.

In both cases, the vegetation contributes to treatment, with greater effect at lighter pollutant loadings. For example, FWS phosphorus removal has been strongly linked to the fractional coverage of different community types (Lakhsman, 1982; Juston, 2006). Therefore, it is useful to distinguish between various degrees of vegetative completeness. Aerial photography or other remote sensing can be used to measure coverage of emergent plants, but it is more difficult to determine the presence of SAV (Rutchey and Vilcheck, 1999). If the wetland has design bathymetry including deep zones, then that information provides estimates of coverage of EAV.

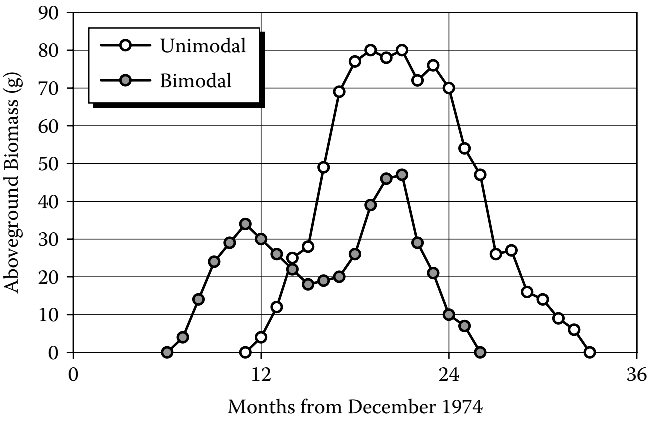


FIGURE 3.13 Growth of single shoots of *Typha domingensis* in a subtropical wetland. (Data from Davis (1989) *Sawgrass and cattail production in relation to nutrient supply in the Everglades*. Sharitz and Gibbons (Eds.). U.S. Department of Energy Conference No. 8603101, held in Charleston, South Carolina; National Technical Information Service: Springfield, Virginia, pp. 325–341.)

Stem Density

The stem density of wetland plants is important because the resistance to water flow is determined in part by stem density. Only a small fraction of the ultimate plant density is planted in a new wetland. Planting densities range from 1,000–40,000 plants per hectare (0.1–4.0 plants per m^2), depending on the rate of spread of the selected plant species and the acceptable timeframe for plant establishment. Through vegetative reproduction, these plants will eventually spread to much greater densities.

Tanner reported 1,400–1,500 stems per m^2 for *Schoenoplectus tabernaemontani* growing in dairy wastewater (Tanner, 2001a), and over 2,000 stems per m^2 for *Schoenoplectus validus* (Tanner, 1994). In contrast, stem counts for *Scirpus acutus* in the Sacramento, California, project were typically only 150 per m^2 in secondary effluent, although accompanied in some cases by 15–30 per m^2 *Typha latifolia* plants (Nolte and Associates, 1997; 1998a).

Cattails generally have many fewer stems per unit area than bulrushes. For instance, the discharge area at Houghton Lake, Michigan, had 71 ± 23 per m^2 for *Typha latifolia*, and 89 ± 22 per m^2 for *Typha angustifolia*. A nutrient-poor location at the same wetland had only 35 ± 22 per m^2 for *Typha latifolia*. Glenn *et al.* (1995) measured 140 per m^2 for *Typha domingensis* in northern Mexico. *Phragmites australis* has comparable numbers in secondary reedbeds, 70–100 per m^2 in the United Kingdom (Daniels and Parr, 1990; as referenced by Cooper *et al.*, 1996). However, *Phragmites australis* grows to higher densities in warm climates, around 250 per m^2 in Australia (Hocking, 1989a).

Hydraulic modeling has therefore adopted similar stem density numbers. For instance, Nepf *et al.* (1997) used stem (cylinders) densities of 200–2,000 per m^2 in constructed flume experiments, to represent *Juncus roemerianus*. Hall and Freeman (1994) studied hydraulics in constructed flumes, with bulrush plants, at densities of 400 and 800 per m^2. In laboratory flumes, Schmid *et al.* (2004b) used 12.8 stems (cylinders) per m^2 as representative of *Typha latifolia*.

Submerged Area

Since microbial transformations within a FWS wetland are largely a function of area available for biofilm growth, the creation of surface area by emergent aquatic plants and associated leaf litter is an important contribution to the treatment process. One method to assess the relative contribution of the plants is to measure the amount of submerged surface area available per area of wetland (submerged specific surface area). For instance, a waste stabilization pond would have a specific surface area of 1.0 m^2/m^2 as the only wetted surface area is the bottom of the pond. Specific surface areas for wetlands are higher, averaging 2.8 m^2/m^2 at depth 30 cm for various species (Table 3.6). The depth dependence of specific surface is nearly linear (U.S. EPA, 1999).

The reader is cautioned that submerged area differs markedly from the leaf area index (LAI), the latter being commonly used in studies of photosynthesis and transpiration. LAI measures the total area of leaves in the air above water. Under most normal depths of operation, the large majority of leaf area will be above water. For instance, *Scirpus* leaves were measured to have LAI of 5.3–6.5 m^2/m^2, and *Typha* of 4.1–5.5 m^2/m^2 at the Sacramento, California wetlands (Nolte and Associates, 1998a).

Underwater Porosity

The actual detention time in a FWS wetland is the wetland water volume divided by the volumetric flow rate. In turn, the actual water volume is less than the bathymetric value, because submerged stems take up space. The literature contains pronouncements of appropriate estimates ranging from 0.65 (Reed *et al.*, 1995) to 0.95 (Kadlec and Knight, 1996). Porosity depends upon stem density and stem size. For cylindrical stems, the relationship is:

$$\varepsilon = 1 - \eta \frac{\pi D^2}{4} \tag{3.2}$$

where
D = stem diameter, m
ε = porosity fraction
η = stem density, no. per m^2

For instance, at the Houghton Lake, Michigan, site, there were 96 *Typha latifolia* stems per m^2, and the mean stem diameter in the 30 cm depth was 1.2 cm. The cylinder porosity was therefore 99%. As may be confirmed from Equation 3.2, it is only when there are large numbers of stems of large diameter that porosity drops below 95%, for example, more than 100 per m^2 at diameter 2.5 cm. Such extreme sizes and densities are uncommon, but may be encountered in warm climates. For instance, Hocking (1989a; 1989b) reports stem densities of 250 per m^2, and basal diameters of one cm may be inferred from his data for *Phragmites australis* in a nutrient-rich warm climate. The corresponding cylindrical porosity is 96%.

In many circumstances in FWS, topographical "blockage" is more important than vegetative wet volume exclusion (see Chapter 2).

TABLE 3.6
Submerged Surface Area in Ponds, and Wetlands at Depth 30 cm

Treatment System	Vegetation	Submerged Area (m^2/m^2)
Waste stabilization pond	None	1.0
Water hyacinth pond	*Eichhornia crassipes*	2.2
Arcata, California	*Scirpus acutus*	4.5
Arcata, California	*Typha latifolia*	2.0
Benton, Kentucky	*Scirpus cyperinus*	3.1
Benton, Kentucky	*Typha latifolia*	2.1
Houghton Lake, Michigan	*Typha latifolia*	2.1
Houghton Lake, Michigan	*Typha angustifolia*	2.7
Pembroke, Kentucky	*Scirpus validus*	2.7
Pembroke, Kentucky	*Typha angustifolia*	3.2

Note: Litter and basin side walls are excluded.

Source: Data from U.S. EPA (1999) *Free water surface wetlands for wastewater treatment: A technology assessment*. EPA 832/R 99/002, U.S. EPA Office of Water: Washington D.C. 165 pp.; and Khatiwada and Polprasert (1999a) *Water Science and Technology,* 40(3): 83–89.

Root Penetration

Early literature on HSSF wetlands contained much emphasis on the importance of root penetration depth and its effect on treatment (U.S. EPA, 1993f; Reed *et al*., 1995). The perception was that some wetland plants would have greater rooting depths, and hence provide more radial oxygen loss to conduct aerobic processes in the rhizosphere. It is indeed true that plants differ in their rooting profiles in relatively clean water, but it is now known that rooting profiles do not differ much among species in nutrient-rich waters (see Chapter 2, Figure 2.29). Roots are predominantly in the upper 20–30 cm of the media in both HSSF and FWS wetlands.

3.3 LITTERFALL AND DECOMPOSITION

Over the life cycle of a vascular plant, all plant tissues are either consumed, exported, or eventually recycled back to the ground as plant litter. Litterfall and the resulting decomposition of organic plant material are ecologically important functions in wetlands, and contribute to the cycling of nutrients and pollutants.

LITTERFALL

Wetland plant tissues fall at variable rates depending on the survival strategy of the individual plant species. Herbaceous plant species typically recycle the entire aboveground portion of the plant annually in temperate environments. The growth season may vary from ten or more months in subtropical regions to less than three months in colder climates. Also, most herbaceous species lose a fraction of living leaf and stem material as litter throughout the growing season, so there is a continuous rain of dead plant tissues throughout the year with seasonal highs and lows of litterfall. Woody plant species also participate in this production of plant litter through a natural pruning of small branches throughout the annual period. In the northern hemisphere, large amounts of flowers are shed during the spring, and leaves and fruiting bodies are lost during the fall.

Most herbaceous wetland plants do not directly fall to the wetland floor after senescence and death. Instead, plants remain in an upright stance until meteorological conditions cause them to topple. Wind, rain, and especially weight of snow, cause the standing dead material to fall. Terminology varies, and so dead material is sometimes called litter, regardless of whether it is upright or not. At other times, a distinction is drawn between standing dead and prone material called litter.

Decomposition

Decomposition generally refers to the disintegration of dead organisms into particulate form (or detritus), and the further breakdown of large particles to smaller and smaller particles, until the structure can no longer be recognized and complex organic molecules have been broken down into CO_2, H_2O, and mineral components (Mason, 1977). In wetland studies, the term decomposition is mostly confined to the breakdown and subsequent decay of dominant macrophytes, which leads to the production of detritus. Most net annual aboveground production of wetlands is not consumed by herbivores but decomposes on the wetland surface. Rates of decomposition vary in wetlands and the fate of materials released and adsorbed during decomposition depends on the physical and chemical composition of material as well as environmental conditions at the site of decomposition (Vymazal, 1995).

Studies of litter decomposition are very numerous in the literature. Techniques for such studies have been compiled in books (Barlocher *et al.*, 2005). Most of these studies have been concerned with aboveground plant parts.

The decomposition of litter and resultant release of nutrients involve at least two processes (Godshalk and Wetzel, 1978a) An initial loss of soluble materials is attributed to abiotic leaching (Boyd, 1970; Gosselink and Kirby, 1974; Godshalk and Wetzel, 1978b; 1978c; 1978d). This process is quite rapid and accounts for the majority of mass reduction during the early stages of decomposition. Leaching occurs very quickly under both aerobic and anaerobic conditions with most of the water-soluble organic substances being released within a few days. The rapid initial release of nutrients by leaching has been documented in many marsh plants—up to 30% of nutrients are lost by leaching alone during the first few days of decomposition (Vymazal, 1995). In submerged and floating-leaved plants, leaching accounts for up to 50% loss of dry matter within the first two to three days. Released nutrients may be incorporated into the protoplasm of decomposer organisms where activities such as respiration and denitrification account for additional nutrient losses (Mason and Bryant, 1975).

Flooding in wetlands has been found to increase the litter decomposition rate through physical leaching of inorganic and organic compounds from the plant tissues (Day, 1989; Whigham *et al.*, 1989) and by providing habitat for aquatic microbes and invertebrates, which are important mediators in this process. However, if flood waters are anaerobic, biological activity is greatly reduced (Tupacz and Day, 1990) and only the leaching mechanisms and anaerobic respiration will occur.

Patterns of Weight Loss

Chemical analysis of plant material reveals different rates of decomposition for different components of the plant material (soluble components, cellulose, hemicellulose, and lignin), and that rates of decomposition of each component change over time, such that the specific rate of decay for each fraction decreases as decomposition proceeds (Moran *et al.*, 1989). The initial sharp drop in necromass is followed by a decline to an undecomposed residual. The initial drop is typically of the order of 10–20% for soft-tissue emergent macrophytes (Table 3.7).

The residual of recalcitrant substances is on the order of 5–20%, as inferred from long-term accretion studies. Rarely are decomposition studies continued to the point where such residuals can be determined. This is in major part due to the length of time required, as well as to the limitations of measurement techniques. An example of a litter residual is shown in Figure 3.14.

When these features are considered in combination, a modified first-order loss equation results:

$$\left(\frac{M - M^*}{M_o - M^*}\right) = A\exp(-kt) \tag{3.3}$$

where

A = fraction remaining after initial leaching
k = mass loss rate coefficient, d^{-1}
M_o = initial mass, g
M = mass remaining, g
M^* = residual mass remaining, g
t = time, d

In the vast majority of literature studies, the value of M^* is chosen to be zero; and the value of A is selected to be unity. There is then only one parameter to consider, the lumped mass loss rate coefficient, and under these special circumstances, it is here denoted by k_1. Chimney and Pietro (2006) provide rates of litter decomposition of 140 different wetland plant varieties (Table 3.8). Mean first-order rate coefficients (k_1) for emergent macrophyte leaf litter decomposition

TABLE 3.7
Initial Weight Loss for Submerged Litter in Treatment Wetlands

Site	Wetland	Water	*Typha*	*Scirpus*	Data Source
Sacramento, California					Nolte and Associates (1998a)
	1A	WW	0.01	0.03	
	1B	WW	0.15	0.35	
	7A	WW	0.03	0.90	
	7B	WW	0.21	0.56	
	9A	WW	0.00	0.82	
	9B	WW	0.17	0.14	
	Mean	**WW**	**0.10**	**0.47**	
Sacramento, California					Nolte and Associates (1998a)
	5A	Control	0.14	0.00	
	5B	Control	0.00	0.00	
	LC3	Control	0.15	0.10	
	LC4	Control	0.18	0.16	
	Mean	**Control**	**0.12**	**0.07**	
Léon, Spain					Alvarez and Becares (2006)
	Winter	WW	0.14	—	
	Summer	WW	0.15	—	
Theresa Marsh, Wisconsin					Puriveth (1980)
	—	Runoff	0.09	0.11	
Houghton Lake, Michigan					Kadlec (1989)
	—	WW	0.14	—	
	—	Control	0.06	—	

Note: WW = wastewater; values were determined by data fitting.

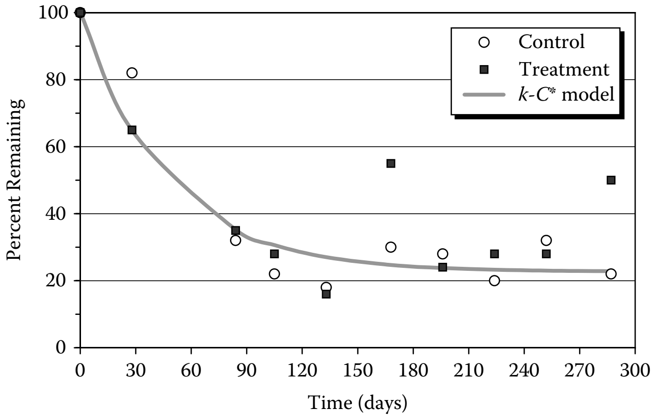


FIGURE 3.14 Leaf litter decomposition in treatment and control wetlands at Thibodeaux, Louisiana. Species were *Fraxinus pennsylvanica, Salix nigra, Taxodium distichum, Nyssa aquatica,* and *Acer rubrum*. Two outliers removed for modeling. (Data from Rybczyk *et al.* (2002) *Wetlands* 22(1): 18–32.)

TABLE 3.8
Summary of Lumped Loss Rate Coefficients for Herbaceous Plants in Various Wetlands

Species	Data Sets N	Mean k_1 (yr^{-1})	Median k_1 (yr^{-1})	Mean Half-Life (d^{-1})
All submersed species	107	17.3	10.2	15
All floating species	80	13.9	8.9	18
All emergent species	280	3.03	0.80	83

TABLE 3.9
Values of the Lumped Loss Rate Coefficients for *Typha* in Various Treatment Wetlands

Species	Location	Mean k_1 (yr^{-1})	Half-Life (d^{-1})
Typha wastewater	Sacramento, California	0.71	356
Typha control	Sacramento, California	0.82	308
Typha wastewater, summer	Leon, Spain	1.57	161
Typha wastewater, winter	Leon, Spain	0.73	347
Typha wastewater, average	Leon, Spain	1.15	220
Typha runoff	Theresa Marsh, Wisconsin	0.70	361
Typha wastewater	Houghton Lake, Michigan	0.50	506
Typha control	Houghton Lake, Michigan	0.71	356
Typha runoff	ENRP, Florida	1.72	147

averaged 1.4 yr^{-1} for 32 studies of *Phragmites australis*, 1.7 yr^{-1} for 23 studies of 10 *Scirpus* species, and 1.4 ± 0.9 for 72 studies of 8 *Typha* species. Variability for a single plant across studies is not great (Table 3.9). The half-life of the litter is equal to $0.693/k_1$.

Litter decomposition is largely mediated by vertebrates, invertebrates, and microbes living in wetlands. New litter is typically conditioned by fungi and bacteria before it is shred into smaller particles by aquatic macroinvertebrates (Merritt and Lawson, 1979). The activity of these organisms is conditioned by temperature, and therefore a temperature effect on decomposition is to be expected. Studies by Alvarez and Becares (2006) confirm this effect, as a differential in rates in summer and winter (Table 3.9). It is also true that warmer climates show higher rates of litter decomposition on an

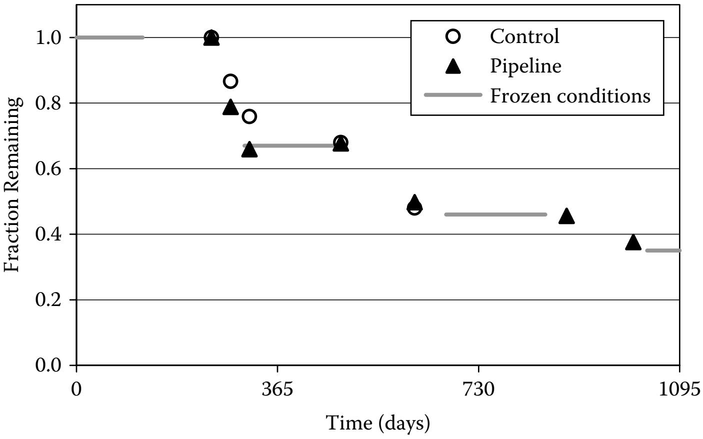


FIGURE 3.15 Decomposition of cattail (*Typha* spp.) litter in wastewater and control areas of the Houghton Lake, Michigan, wetland. Material was placed on September 2. Freeze-up occurs around November 1, and thaw around May 1. No weight loss occurred during frozen conditions.

annual basis. However, the effect of frozen winter conditions typically interrupts the decay processes, which effectively come to a halt in frozen water and soils (Figure 3.15). Therefore, part of the variability across data sets has to do with this winter-season shutdown.

Combined Effects of Successive Cohorts

Research has in general focused on the fate of a particular cohort of necromass, placed in a porous bag and isolated from other materials in the wetland. However, the litter layer in the wetland is the result of many such cohorts that accrue over the years, and the decomposition processes that reduce each of them over time. A conceptual model of this successive accrual and decomposition is shown in Figure 3.16, for the case of startup of a new wetland. As a simple example, consider litter which has a half-life of one year, being

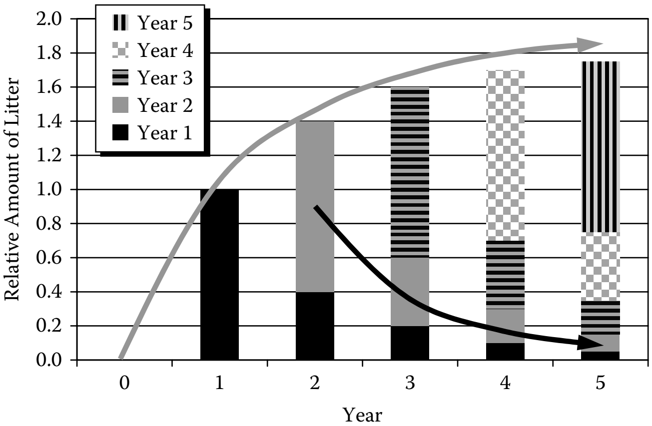


FIGURE 3.16 Cohorts of litter accumulate and decompose during the course of time. (From Rybczyk *et al.* (2002) *Wetlands* 22(1): 18–32. Reprinted with permission.)

deposited once per year in cold temperate climate. At the end of year one, a fresh "crop" of litter of mass M_o is present. At the end of year two, half that remains, and another crop of M_o is added, with the total now being 1.5 M_o. A bit of arithmetic shows that, after a period of some years, this process will lead to an end-of-season litter crop that is twice the annual litterfall. It will take five years to build the litter to 97% of the final value. Of course, events are not so simple in a real situation, but this conceptual model serves to illustrate that a wetland has considerable "memory" via the process of litter accumulation and decomposition.

Belowground Decomposition

Roots and rhizomes also undergo mortality and decomposition. Asaeda and Nam (2002) found a mean half-life of 1.2 years for *Phragmites* rhizomes of age greater than one year. Hill (1987) found 1.84 years for below ground cattail (*Typha angustifolia*). Sharma and Gopal (1982) reported 75% loss of *Typha elephantina* rhizomes decomposed in six months, in India (half-life 0.25 years). Tanner (2001a) found 1.5–2.0 years half-life for rhizomes of soft stem bulrush (*Schoenoplectus tabernaemontani*). Prentki *et al.* (1978) reported 1.5–2 years for *Typha* rhizomes and at least three years for *Phragmites* rhizomes. The fraction of this necromass which contributes to below ground soil accretion has not been determined. It seems probable that most root-rhizome necromass is recycled and only a small fraction ultimately contributes to an underground residual soil accretion. However, the rates of decomposition are slower than for aboveground litter, and therefore the belowground litter crop is much more than double the annual belowground production. It also takes much longer for the belowground litter standing crop to develop.

Thatch

In especially hot and arid climates, treatment wetlands can accumulate excessive quantities of dead plant biomass,

FIGURE 3.17 Thatch at the Tres Rios Hayfield wetland near Phoenix, Arizona. This standing dead material is over two meters in height, and has totally blocked light penetration to the wetland water.

regionally referred to as *thatch*. This accumulation results from the low decomposition rates occasioned by lack of water to support decomposer organisms, plus the upright orientation of the necromass, which keeps the material in the air rather than in the water. The high productivity of the litter, coupled with slow decomposition, leads to very large standing crops of standing dead thatch (Figure 3.17). Mechanical harvesting may be used to remove standing dead aboveground material (combing or thinning) or both dead above- and belowground (thatching) (Nolte and Associates, 1998b; Thullen *et al.*, 2002). Controlled burning is one alternative to remove excess plant biomass in wetland treatment systems, although ash produced by burning will reintroduce nutrients into the water column. This can potentially cause a short-term decrease in treatment efficiency. Burning has been implemented at sites that permit such activities. Since accumulated plant necromass can regenerate, the benefits of removal are only temporary (Thullen *et al.*, 2002).

In contrast, in cold climates the presence of standing dead material provides an excellent adjunct to insulation. Firstly, the standing material protects the wetland soil or water surface from direct exposure to the wind. This wind-break function is probably secondary to the function of catching snow, often to a depth of a meter or more. Snow is held up on dead plants, creating a zone of air spaces interlaced with plant material and captured snow (Figure 3.18). This composite is an excellent thermal insulator, and often prevents freezing in vegetated natural wetlands at times when water is deeply frozen. This function is served for both FWS and HSSF wetlands.

The litter layer on top of a HSSF wetland bed functions as mulch. Such a layer also provides air spaces and holds up the snow to form an insulating layer for the SSF bed.

Mineral Constituents of Litter

The chemical composition of litter is not fixed during decomposition. Carbon and macronutrients (N, P, Ca, K) may be depleted or amplified at differential rates. Decomposer

FIGURE 3.18 Standing dead wetland plants capture snow and provide thermal insulation.

organisms utilize chemicals from both water and the litter, and then contribute to the overall biomass of the litter. For example, the rate of concentration increase may exceed the rate of necromass loss, thereby creating an increase in the mass of a constituent (Figure 3.19). The additional chemicals are acquired from the wetland water. In other situations, there can be a mass loss of chemicals accompanying the loss of necromass (see, for instance Kulshreshtha and Gopal, 1982; Corstanje *et al.*, 2006). There appears to be no universal pattern for the time series of litter chemical composition in natural or treatment wetlands (Chimney and Pietro, 2006).

Accretion

Wetland ecosystems are often sites of long-term positive net primary productivity (NPP), and develop accumulations of buried organic matter in the form of peat and eventually coal. This net accumulation of organic matter is primarily because of the reduced metabolic rate of microbes in flooded wetland sediments compared to metabolic rates in well aerated, upland soils. When living and dead plant material sinks to the level of anaerobic sediments, it is protected from abundant free oxygen and from the higher rates of degradation typical of an oxygenated system.

Therefore, not all of the dead plant material undergoes decomposition. Some small portions of both aboveground and belowground necromass resist decay, and form stable new accretions. The amount of such accretion has been quantified in only a few instances for free water surface wetlands (Craft and Richardson, 1993; Reddy *et al.*, 1993; Rybczyk *et al.*, 2002), although anecdotal reports also exist (Kadlec, 1997a; Sees, 2005; Wang *et al.*, 2006a). Quantitative studies have relied upon either atmospheric deposition markers (radioactive cesium or radioactive lead) or introduced horizon markers, such as feldspar or plaster. Either technique requires several years of continued deposition for accuracy.

The manner of accretion has sometimes been presumed to be sequential vertical layering (Kadlec and Walker, 1999; Rybczyk *et al.*, 2002), but that view is likely to be overly simplified. At least two factors argue against simple layering:

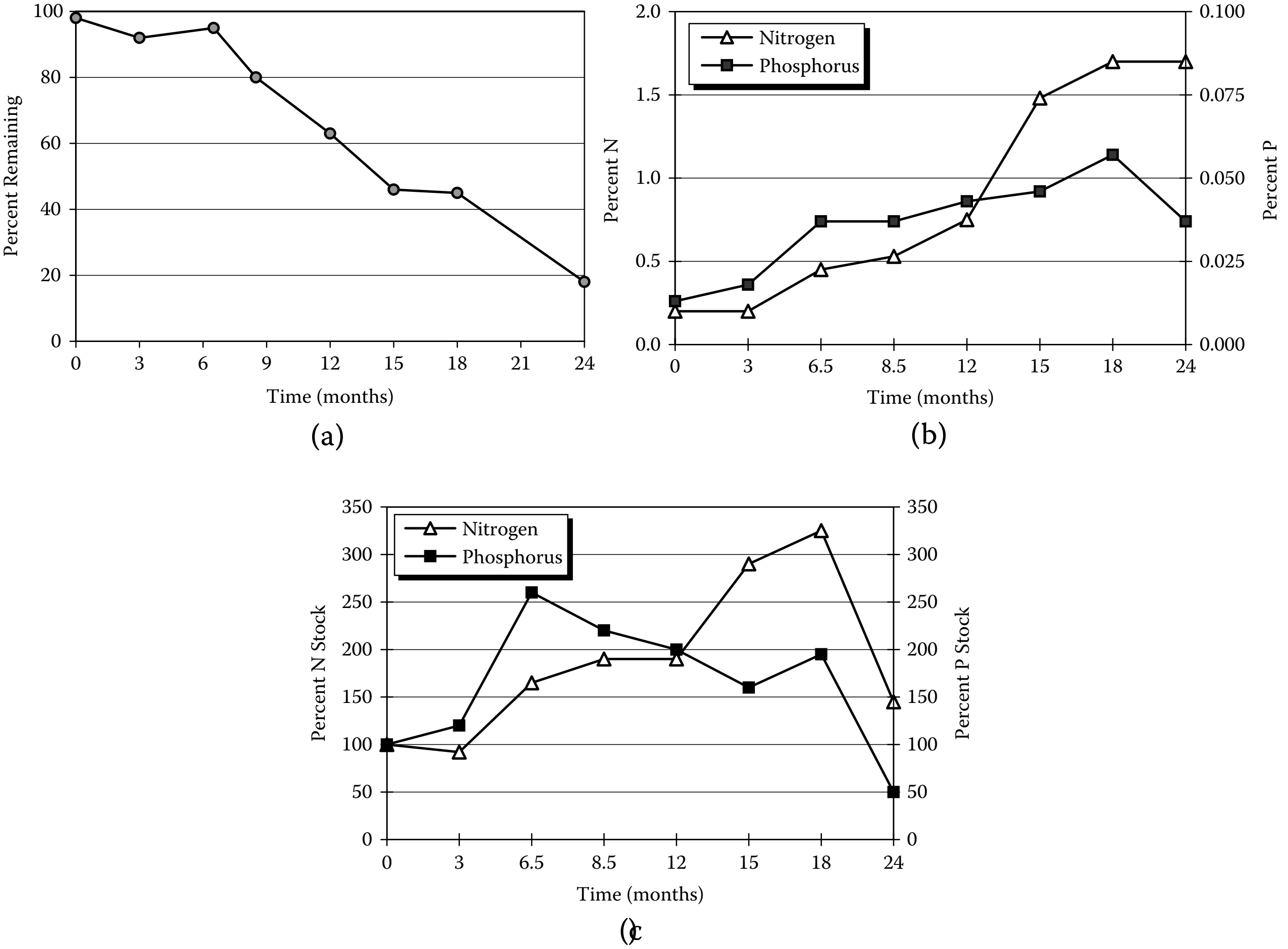


FIGURE 3.19 Changes in amount of culm litter (a), nitrogen (N), and phosphorus (P) content (b), and N&P stock (c) for *Phragmites* over a two-year period of decomposition. (From Gessner (2000). *Aquatic Botany* 66(1): 9–20. Reprinted with permission.)

vertical mixing of the top soils and sediments (Robbins, 1986), and the injection of accreted root and rhizome residuals at several vertical positions in the root zone. Nonetheless, new residuals are deposited on the wetland soil surface, from various sources. The most easily visualized is the litterfall of macrophyte leaves, which results in top deposits of accreted material after decomposition. However, algal and bacterial processing that occurs on submersed leaves and stems results in litterfall and accretion of micro-detrital residuals.

The net result of undecomposed residuals is the buildup of new sediments and soils in the treatment wetland. These residuals are composed of both undecomposed plant parts and the remains of organisms that have caused the decay. The rate of such buildup is often in the range of 0.1–2.0 cm/yr.

Background Concentrations

Wetland systems are dominated by plants (autotrophs), which act as primary producers of biomass. However, wetlands also include communities of microbes and higher animals, which act as grazers (heterotrophs) and reduce plant biomass. Most wetlands support more producers than consumers, resulting in a net surplus of plant biomass. This excess material is typically buried as peat or exported out of the wetland (Mitsch and Gosselink, 1993). This net export results in an internal release of particulate and dissolved biomass to the water column, which is measured as nonzero levels of BOD, TSS, TN, and TP. These wetland background concentrations are typically denoted by the term C^*. Enriched wetland ecosystems (such as those treating wastewater) are likely to produce higher background concentrations than oligotrophic wetlands because of the larger biomass cycling resulting from the addition of nutrients and organic carbon. Even land-locked wetland basins, which only receive water inputs through precipitation, will have nonzero background concentrations. Rainfall and dryfall contain these same substances, and therefore contribute to background concentrations.

Background concentrations are achieved when wetland inflows and outflows contain the same (low) levels of constituents. That situation typically occurs far from the inflow sources of those compounds for flow through systems, and at long times for batch systems exposed to doses of the compounds. Because of random wetland processes, background concentrations may fluctuate markedly around a mean time average value. Atmospheric deposition, uptake, and return processes are in balance (Figure 3.20). The first-order areal model for pollutant removal will be described in detail in Chapter 6, but here the ramifications of decomposition processes are briefly explored. The mass balance for background conditions is:

$$QC_i - QC_o = 0 = (kC^* - R - PC_p)A \tag{3.4}$$

where

A = wetland area, m^2
C^* = wetland background concentration, mg/L
C_p = atmospheric deposition concentration, mg/L
C_i = inlet concentration, mg/L
C_o = outlet concentration, mg/L
k = removal rate coefficient, m/d
Q = flow rate, m^3/d
P = rain rate, m/d
R = return rate from decomposition, g/m^2·d

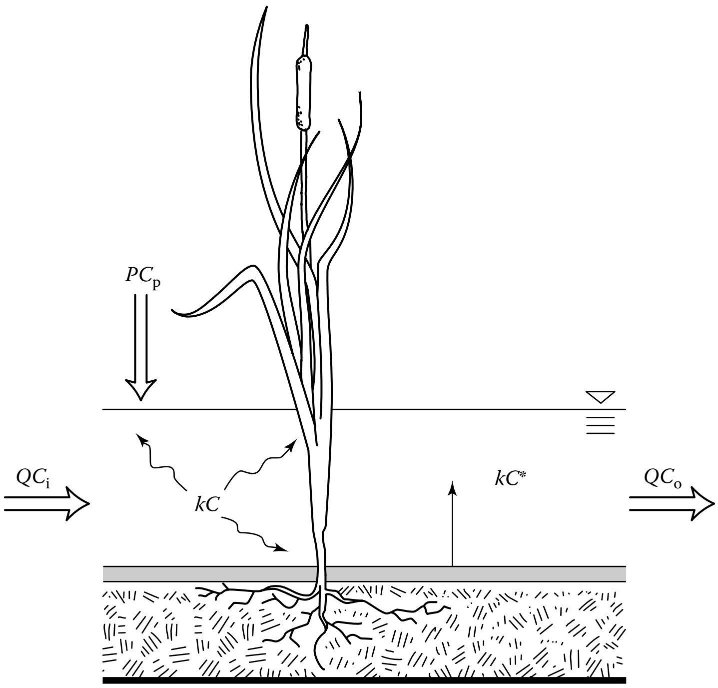


FIGURE 3.20 The background concentration is determined by processes far from inflow effects in a flow through wetland. In that situation, $C_i = C = C_o$.

As a result, the background concentration is that required for a balance between uptake and the combination of atmospheric deposition and return flux from decomposition:

$$C^* = \frac{(R + PC_p)}{k} \tag{3.5}$$

The return fluxes for dissolved organics (BOD) and organic nitrogen are often quite large, and result in $C^* \approx 5$ mg/L and 1.5 mg/L, respectively. On the other hand, phosphorus, nitrate and ammonia are utilized by a variety of biota, and uptake often far exceeds the return flux, resulting in $C^* \approx 0$ mg/L. These values, and methods for determination, will be discussed in more detail in later chapters, by compound.

Wastewater Stresses

Plants living in FWS and SSF treatment wetlands may be subjected to a different set of conditions than plants in natural wetlands. If the application is for domestic wastewater polishing, the incoming water quality is often as good or better than most natural wetlands. The same is true for many remediation applications, in which the chemical targets do not particularly influence nutrients or wetland biogeochemical cycling. Likewise, applications for drinking water conditioning, and crop and urban runoff treatment, do not push the boundaries of wetland water quality environments. Even if the water quality is nonthreatening, treatment wetlands have water level controls, which may be inadvertently set at water levels that are detrimental to the selected or existing wetland plants. Many wetland plants prefer water depths of less than 40 cm, and most also prefer intermittent rather than continuous flooding. Relatively stable water levels, rather than seasonal and rain-driven hydrologic regimes, may place stress on wetland vegetation. The hydrologic requirements of wetland plants are a design consideration (see Part II).

However, treatment of primary domestic wastewaters, food and animal waste, acid mine waters, and leachates, and sludge consolidation, all may create unusual and stressful water quality conditions for wetland plants. The conditions that may be created by strong wastewaters include:

- High influent oxygen demand, which leads to reducing conditions (low redox potential) in the water column and in the wetland root zone
- High nutrient loadings, which lead to increased production of plant biomass and detritus, and subsequently to a high internal oxygen demand
- High sulfur, leading to sulfide toxicity
- Extraordinarily high or low pH
- High salinity, created by large dissolved salt concentrations

Oxygen Deficiency

Under primary or secondary domestic wastewater loading, the influent BOD, nitrogen, and phosphorus are typically much higher than in natural wetlands. Due to the additional oxygen demand from the wastewater, there is generally little or no dissolved oxygen in the FWS water column. The nutrient loadings increase biomass production, which in turn increases the amount of decaying plant material in the detritus layer. These two effects create a strongly reducing (highly anaerobic) sediment layer, and anaerobic soils beneath. The chemical gradient between the oxygen in the root tissue and the sediment is greater, leading to increased oxygen losses from the root tissue (Sorrell and Armstrong, 1994; Cronk and Fennessy, 2001). Wetland plants may develop a thick, waxy coating on mature root and rhizome tissue. However, on the newly growing root hairs (especially at the root tip), oxygen can be easily transferred from the root to the sediment due to the thinness of the cell walls.

Wetland plants attempt to minimize this oxygen loss by preferentially rooting in the uppermost sediment layers, where the least reducing conditions are present (Lockhart, 1999). Under extreme conditions, rooting may preferentially occur in the water (adventitious roots). Under oxygen deficiency, emergent plants can tolerate less flooding; typically the maximum allowable water depth for a given plant species subjected to wastewater loading is less than half of that for the same species in an oligotrophic wetland environment.

Plants living in HSSF wetlands are subjected to stresses similar to FWS wetlands, but additionally possess a relatively hostile rooting environment. Unless very fine sands or soils are used, the capillary action and moisture holding capacity of the bed media is much less than that of natural wetland sediments. Plant root networks must be submerged in order to survive (submersion is especially important during plant establishment). For HSSF systems receiving primary (septic tank) effluent, a strongly reducing (highly anaerobic) environment will develop in the bed matrix. The required nutrient supply is overabundant, and extensive, deep rooting is not necessary to acquire nutrients. Wetland plants respond by preferentially rooting in the uppermost bed layers and by reducing the overall root biomass (Lockhart, 1999). This limited root penetration can create preferential flow paths through the lower section of the gravel bed (Breen and Chick, 1995; U.S. EPA, 2000a; Whitney *et al.*, 2003). Root penetration to the bottom of the bed is likely to occur only in systems that receive low-oxygen demand waste (e.g., a nitrified influent), or have some other means of supplemental oxygen transfer (Behrends *et al.*, 1996; Lockhart, 1999).

Sulfide Toxicity

Lamers (1998) documents that sulfate has negative effects on the growth rate of *Carex nigra*, *Juncus acutiflorus,* and *Gallium palustre*, at concentrations of 64 and 128 mgS/L. Koch and Mendelssohn (1989) report that 32 mgS/L of sulfide produced negative effects in *Panicum hemitomon* and *Spartina alterniflora*. The presence of sulfide is coupled with anaerobic conditions in the root zone, but the effects of sulfide go beyond mere anoxia (Koch and Mendelssohn, 1989). Hydrogen sulfide apparently inhibits the activity of alcohol dehydrogenase, thereby limiting the ability of plants to avail themselves of alternative anoxic energy pathways. This effect was confirmed by measuring a reduced ^{15}N uptake rate in the presence of sulfide. However, the availability of free sulfide is strongly mediated by the presence of iron, because of the formation of iron sulfides.

Phytotoxicity was found to be very serious at the 45 mgS/L level in *Phragmites australis* (Armstrong *et al*., 1996). These authors found that aeration pathways became blocked, interfering with the diffusive connection to the atmosphere, and thus reducing the plant's ability to oxygenate the rhizosphere. Smolders and Roelofs (1996) found, for *Stratiotes aloides*, an aquatic macrophyte characteristic of mesotrophic freshwater marshes, that levels of 320 mgS/L were toxic to the roots. Lamers *et al.* (2002) found root parts growing in 1.7–3.4 mgS/L of sulfate into the peaty sediment, clearly showed sulfide toxicity by becoming black, slimy, and unfit for nutrient uptake from the sediment. Free sulfide could not be detected in the surface water. They concluded that only roots in the surface water would survive. *Nuphar lutea* did not propagate in the sulfate-treated enclosures. However, the sensitivity of a wetland plant species to free sulfide not only depends on the actual sulfide levels in the rhizosphere, but also on detoxification mechanisms such as radial oxygen loss.

Extreme Salt Content and/or pH

Acid mine drainage wetlands often operate with incoming pH less than 5, which is commonly regarded as a lower limit for aquatic resource protection (U.S. EPA, 2006), and pH 6.5 is preferred. Although there are many plants that can tolerate low pH, the diversity of treatment wetlands operating under extremes will be constrained. Indeed, natural northern bogs commonly have pH less than 5, as a result of the decomposition processes and conditions that prevail. Likewise, high pH is found in other situations, such as leachates from waste material piles from the phosphate and soda ash industries, and from construction debris. There are natural wetlands with high pH, including prairie potholes and playas in the United States. Again, there are many plants that can tolerate high pH, but the selection for these alkaline treatment wetlands will be limited.

There are major differences between the species of plants that inhabit saline wetland environments and those that live in freshwater wetlands. Treatment wetlands are almost always utilized for fresh waters, but high salt content is sometimes a feature of the incoming water. Species such as *Typha* and *Phragmites* are tolerant of a wide range of salinity, and will do well in environments with high TDS. However, some plants normally inhabit saline or brackish water, including, for example, *Spartina* spp. (cordgrasses) and *Juncus maritimus* (seaside rush). The reader is referred to the vast literature on the characteristics of salt marsh plants if a high salinity treatment wetland is contemplated.

3.4 VEGETATIVE COMMUNITIES IN TREATMENT WETLANDS

ALGAL SYSTEMS

Periphyton

Natural Everglades periphyton-dominated wetlands exist and function at phosphorus levels below 10 ppb. Constructed wetlands dominated by periphyton, termed periphyton stormwater treatment areas (PSTAs), have also been successful in closely approaching the 10 ppb goal in small units. Periphyton-based STAs (PSTA) and submerged aquatic vegetation (SAV) wetlands are variants on the same theme: shallow submersed aquatic vegetation that supports an active periphyton community. Both envision sparse emergent vegetation that forms an anchor and a substrate for the periphyton. Emergent vegetation must be very sparse to avoid shading of the algae, which occur in three forms: on the bottom, as floating mats, and as attached growth on submerged plant parts (Figure 3.21). The benthic mats can access residual phosphorus in the sediments and recycle accreted phosphorus. PSTA envisions sparse vegetation that forms an anchor and a substrate for periphyton. Emergent vegetation must be very sparse, if present at all, to avoid shading of the algal mats, which occur on the bottom as floating mats, and as attached growth on submerged plant parts. Accretion of residuals is needed to make this a passive sustainable process. The benthic mats can access such residuals and recycle accreted phosphorus.

It should be recognized that periphyton treating water of concentration greater than about 10 ppb would not be pristine Everglades periphyton. Extensive research has shown that pristine cyanobacterial mats do not survive at concentrations above that limit. That research shows that at higher concentrations, the periphyton contains a significant proportion of green algae. At some higher phosphorus concentration,

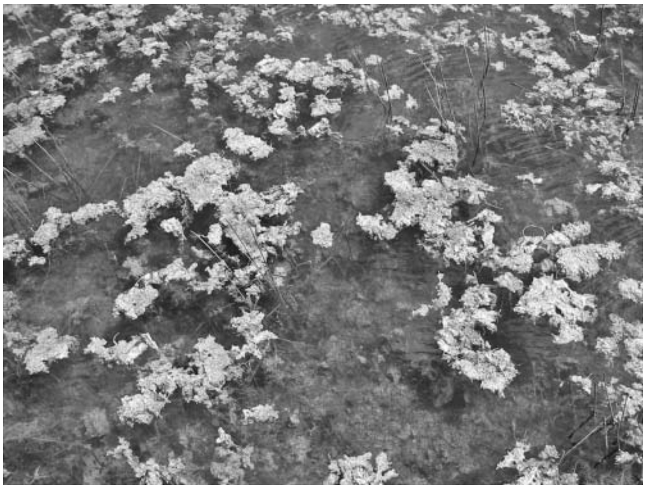


FIGURE 3.21 A periphyton-dominated wetland contains sparse emergents, for protection and anchoring. The algal mats may be either floating or resting on the bottom.

approximately 50 ppb, the existence of any kind of self-sustaining, algal-dominated system is threatened.

There have been eleven constructed projects in South Florida; and, supplemented by natural system response studies, form an impressively large suite of datasets (Kadlec and Walker, 2003). The two largest of these constructed systems are 40 ha in extent.

Algal Turf Scrubbers

Algal turf scrubbers are channels with shallow water flow, vegetated by filamentous algae. These have been utilized in Asia (Kim *et al.*, 2002), Europe (Schumacher and Sekoulov, 2002), and North America (Adey *et al.*, 1996; Craggs *et al.*, 1996a; 1996b). The performance of algal biofilm processes is comparable with suspended algae systems. The algae grow rapidly in nutrient rich water, and adhere to available surfaces. Harvest is a necessity, else the biomass begins to slough, and effectiveness is lost. Therefore, the success of this technology is very much dependent upon the infrastucture used to support the organisms. The organisms may include individuals or mixtures of green algae (*Stigeoclonium, Oedogonium, Ulothrix, Scenedesmus*, *Spirogyra*), blue-green algae (*Oscillatoria, Lyngbya*), and diatoms.

Submerged Plants

Submerged aquatic vegetation (SAV) such as waterweed (*Elodea* spp.), coontail (*Ceratophyllum* spp.), and naiads (*Najas* spp.) have been used to treat wastewater (Gumbricht, 1993a; 1993b). These submerged plants have parts suspended in the water column, and are sometimes rooted in the bottom sediments. Typically, their photosynthetic parts are in the water column, but certain species may grow to where their photosynthetic parts are at or just above the water surface. This category of constructed FWS wetland has not had widespread usage, but submerged plant species are present in many natural treatment wetlands, and are invaders in other constructed wetlands.

Examples are presented here to illustrate usage of this type of treatment wetland vegetation.

Secondary Wastewater Treatment, Australia

An experimental trench, 4 × 100 m, was established and used for two years (1984–1986) to treat secondary municipal water (Bavor *et al.*, 1988). The trench contained 100% cover of parrot feather (*Myriophyllum aquaticum*). Parrot feather is regarded as a "mostly" submersed plant (Collins *et al.*, 2005), but with floating parts under some circumstances. Four other trenches contained emergent plants in varying proportions. The *Myriophyllum* trench had the poorest performance.

River Treatment, Sweden

The submersed macrophyte treatment system at Snogeröd, Sweden was put in operation in 1988, and operated until 1991 (Gumbricht, 1993a). The 1.2-ha wetland contained *Elodea canadensis* and *Cladophora glomerata*, and was operated at a depth of 0.6 m and a flow of 2,400 m^3/d. The incoming water had TN = 9.8 mg/L and TP = 0.26 mg/L; wetland effluents averaged TN = 7.5 mg/L and TP = 0.07 mg/L. The conclusion from this field-scale project was that submersed macrophyte systems have the potential of polishing river waters and pretreated wastewaters. Gumbricht (1993b) went on to speculate that harvest could be used to improve performance.

Municipal Wastewater Polishing, Netherlands

A treatment wetland system on the island of Texel in the Netherlands was constructed in 1994 to polish 3,400 m^3/d of effluent from a 45,000 PE municipality (Toet *et al.*, 2005). The surface flow wetland had a total water surface of 1.3 ha. The STP effluent first entered a presettling pond, was then divided over nine parallel ditches, after which it was collected in a discharge ditch and discharged to surface water. The first half of eight ditches was 0.2 m deep and contained *Phragmites australis* or *Typha latifolia*, while the second half was 0.4 m deep and contained submerged aquatic macrophytes (*Elodea nuttallii*, *Ceratophyllum demersum*, and *Potamogeton* spp.). The SAV portion of the system removed essentially no phosphorus, because of high loading rates, but did reduce nitrogen by 45%.

Agricultural Runoff, Florida

Cell 4 of the Everglades Nutrient Removal Project (ENRP) was a 147-ha constructed wetland that developed into a SAV system, by virtue of herbiciding competing emergents (Figure 3.22). Emergents were spot-sprayed, which required relatively small quantities of chemicals. For example, the sum of all herbicide applications in 1998 averaged only 3.0 L/ha (SFWMD, 1999a; Dierberg *et al.*, 2002). The use of herbicides as a feature of treatment wetland operation and maintenance is perhaps unique to south Florida, and was exercised for the first time in the ENRP.

Recognizing the good performance of SAV in cell 4, phosphorus removal was investigated in mesocosms stocked with a mixture of taxa common to the region: *Najas guadalupensis*, *Ceratophyllum demersum*, *Chara* spp. and *Potamogeton illinoensis* (Dierberg *et al.*, 2002). After eight months of operation, *N. guadalupensis* dominated the standing crop biomass and phosphorus storage. The mean inflow TP concentration of 107 μg/L was reduced to 52, 29, and 23 μg/L in the 1.5, 3.5, and 7.0 day HRT treatments, respectively.

As a result of these research and demonstration projects, SAV was specified for the outlet sections (about 50%) of all the stormwater treatment wetlands (STAs), by virtue of state law. Conversion of the outlet wetland cells is underway at the time of this writing, and been completed in large measure. Approximately 8,000 ha of SAV constructed wetlands will result.

Floating Plants

Floating aquatic vegetation (FAV) treatment systems consist of one or more ponds in which one or more species of water tolerant, floating vascular plants are grown. The shallower depths and the presence of floating aquatic macrophytes

FIGURE 3.22 Coots enjoy the submerged aquatic vegetation in Cell 4 of the Everglades Nutrient Removal Project, a 147-ha constructed wetland in Florida.

in place of algae are the major differences between these aquatic treatment systems and stabilization ponds. The presence of plants is of great practical significance because the effluent from aquatic systems is often of higher quality than the effluent from stabilization pond systems with no floating plants, for equivalent detention times. Floating aquatic plant wetlands are described in detail by DeBusk and Reddy (1987), Crites and Tchobanoglous (1998), and Crites *et al.* (2006). Their major application has been in the tropics and subtropics.

In FAV systems used for municipal wastewater, the carbonaceous biochemical oxygen demand (CBOD) and suspended solids (SS) are removed principally by bacterial metabolism and physical sedimentation. In systems used to treat CBOD and SS, the plants themselves bring about very little actual treatment of the wastewater. Their function is to provide components of the aquatic environment that improve the wastewater treatment capability and/or reliability of that environment. In aquatic treatment systems designed to remove nutrients (N and P), plant uptake can contribute to the removals, especially where plants are harvested frequently.

The principal floating plant species used in aquatic treatment systems are water hyacinth (*Eichhornia crassipes*), pennywort (*Hydrocotyle* spp.), and duckweed (*Lemna* spp.). These, and other floating species such as water lettuce (*Pistia stratiotes*) and mosquito ferns (*Azolla* spp.), may occur in any FWS wetland. Water hyacinths have been used in a variety of experimental and full-scale systems for treating wastewater (see, for instance, Reddy and Smith, 1987). The use of water hyacinths has been limited in geographic location to warm weather regions because of the sensitivity of water hyacinth to freezing conditions. Duckweed systems have been developed in colder climates because of the greater temperature tolerance of duckweed species. Both duckweed and water hyacinth systems have most often been used for either removing algae from oxidation pond effluents or for nutrient removal following secondary treatment.

Floating plants have their photosynthetic parts at or just above the water surface with roots extending down into the water column. Nutrients are taken up from the water column through the roots. These roots provide an excellent support medium for the growth of bacteria and for the filtration/adsorption of SS. Root development is a function of nutrient availability in the water and growth rate of the plant. Thus, in practice, the density and depth of plant roots will be affected by pretreatment, and by other factors affecting plant growth rate such as temperature and harvesting. With floating plants, the penetration of sunlight into the water column is reduced and the transfer of gas between water and atmosphere is restricted. As a consequence, floating plants tend to keep the wastewater nearly free of algae and anaerobic or nearly so.

In this book, designed FAV systems are regarded as a modification of pond or lagoon treatment technology, rather than a variety of wetland. That appears to be the decision of much of the literature on FAV systems; however, some authors classify them as wetlands, for instance, Nahlik and Mitsch (2006). FAV systems are an alternative to FWS emergent marshes and SSF systems under appropriate circumstances. They provide a better opportunity for harvesting, but are difficult to maintain in large cells.

Floating plants can invade FWS wetlands that were not designed to include such vegetation, and therefore some examples are included here.

Volunteer FAV

Floating plants are easily able to invade open water zones of FWS wetlands. Systems in the southern United States, for example, are susceptible to larger plants, such as water hyacinths (*Eichhornia crassipes*), water lettuce (*Pistia stratiotes*), and pennywort (*Hydrocotyle* spp.), while northern systems typically experience duckweed (*Lemna* spp.).

Duckweed

Probably the most common floating plant in constructed treatment wetlands is duckweed (*Lemna* spp.). It colonizes with great ease, and is geographically widespread. It has been advocated as a treatment system in itself (see, for example, Smith and Moelyowati, 2001; Körner *et al.*, 2003; Ran *et al.*, 2004), but requires a retaining grid to prevent wind-driven drift of the plants. Small patches of open water in FWS wetlands, such as intentional deep zones and muskrat eat-outs, are very often covered with duckweed (Figure 3.23).

Pennywort

The Cobble wetlands at Tres Rios, Arizona, passed through a startup period of approximately one year, after which they operated in a stable vegetation mode for approximately two years. But in the spring of 1998, the planted wetland vegetation (bulrushes) died entirely (Kadlec, 2006b). The cause of the demise of the selected plants is not definitively known. Subsequently, the wetland was reconfigured as braided channels connecting former deep zones. The wetland then underwent a period of regrowth, and floating aquatic plants colonized, and pennywort (*Hydrocotyle ranunculoides*), eventually creating near-complete cover (Figure 3.24).

Water Hyacinths

In warm climate FWS wetlands, water hyacinths and water lettuce are ready invaders of open water areas. For example, water hyacinths invaded most of cell 1B of STA5 of the Everglades Protection Project wetlands, and overgrew the intended SAV in the 494-ha wetland. As suggested above, hyacinths are not necessarily a bad alternative for treatment, but physical problems occurred in this case. Because of the large size of the cell, the wind fetch was an unobstructed three kilometers. Even a modest westerly (prevailing) wind caused the floating hyacinths to drift to the outlet, and jam outlet structures. A strong wind caused vegetation to pile up to depths of about a meter, half abovewater. These windrows created water backup, and badly interfered with hydraulic operation of the wetland. The hyacinths were controlled by herbiciding (SFWMD, 2004).

Floating Mats and Rafts

Floating islands or mats are widespread vegetation formations that occur in all climatic regions of the globe (van Duzer, 2004). These range in character from the floating sedge fens of Alaska (Racine and Walters, 1991) to the papyrus swamps of equatorial Africa (Gaudet, 1977; Kansiime and Nalubega, 1999). For example, very large areas of the Mississippi River delta wetlands are floating mat systems (Sasser *et al.*, 1996), comprising over 70% of the western Terrebonne Basin.

There are at least three natural formation mechanisms (Clark and Reddy, 1998):

1. The delamination and floating of unvegetated organic substrates from deeper sediment. Germination of plants occurs after emergence. This is a peat float-up process.
2. The rhizomes of aquatic plants colonize the water surface from a nucleus of aquatic vegetation that is either unattached or expanding from the shore. This is the grow-over process.
3. Units of rooted vegetation and substrate split simultaneously from the bed, and float to the wetland surface. This is a mat floating process.

Floating mats must be almost entirely organic in order to be buoyant enough to float. They derive their buoyancy from gas spaces in rhizomes (Hogg and Wein, 1987; 1988; Krusi and Wein, 1988), and also from gases generated by decomposition processes. However, floating plant mats may also be artificially fostered in aquatic or wetland systems, by use of rafts of one sort or another.

A distinction is drawn between treatment systems that contain floating plants and those that contain floating mats. If the plants can normally float as individuals, without any

FIGURE 3.23 *Lemna* filled all open water areas in the Lake Nebagamon, Wisconsin, constructed wetland during the startup grow-in period.

FIGURE 3.24 An unintended cover of pennywort (*Hydrocotyle* spp.) in Cobble Cell 1 (0.92 ha) at Tres Rios, Arizona.

support from a substrate or their neighbors, then the system is a floating plant system. Well-known examples of constructed floating plant systems are water hyacinths (*Eichhornia crassipes*), water lettuce (*Pistia stratiotes*), duckweed, (*Lemna* spp.) and water fern (*Azolla* spp.). In contrast, a very much larger category of plants may be established in floating systems in which supporting media and neighbors are required. A total of 67 different plants have been tested in Hungary (Lakatos, 1998).

Here we are concerned about two aspects of floating mat systems: their unintended development in treatment wetlands, and the intentional design of floating plant mats for wastewater quality improvement. Interestingly, such systems are not often considered as a constructed wetland design option.

Unintended Floating Mats in Treatment Wetlands

In several instances, treatment wetlands have developed floating mats of vegetation, which were unplanned and unexpected. A few illustrative examples are given here. Treatment has sometimes continued to be effective; but has been impaired in other cases. Of course, the water flows under such mats, rather than over and through the litter layer. This is a major difference from the common marsh–overland flow system. Floating mat systems may be more akin to floating plant systems, such as water hyacinths, but no direct comparisons have been done.

Kis-Balaton, Hungary

The constructed shallow treatment impoundments (wetlands) on the Zala River, as it enters Lake Balaton, function for nutrient removal. The original vegetation of the second unit consisted mainly of reed beds (*Phragmites*). As a consequence of routing the river, the reed beds were damaged. Dead rhizomes produced gas, which buoyed fragments of the reed bed to the surface as a floating mat. These mats formed a matrix for secondary succession. These floating islands are partly attached to the still-living fragments of reed beds rooted in the sediment, and have an approximate diameter of about 15 m. The approximate rhizome mat thickness of the floating islands was 0.5 m in 2001. The islands were characterized by willows (*Salix cinerea*), sedges (*Carex* spp.), cattail (*Typha angustifolia*), and ruderal species (e.g., *Bidens cernuus*) (Somodi and Botta-Dukat, 2004).

Kinross and Houghton Lake, Michigan

These two natural wetlands developed near-monocultures of cattails on preexisting peatlands. Over the course of time, these *Typha* communities became floating mats (Kadlec and Bevis, 1990). A possible cause was the retreat of the root zone to a smaller biomass located high in the soil profile, compared to prior conditions. This physical effect, coupled with possible partial peat dissolution into the less acidic added wastewater, led to a 50-cm soil-free water zone topped by the floating mat. The mats are closely woven beds of roots, rhizomes, and sediments (Figure 3.25). These had enough strength to permit foot and small all-terrain vehicle travel on the mat. Treatment continued to be generally effective under the mat, except that early phosphorus additions to the Kinross system were later in a bleed-back mode (see discussion on woody plants)

Lake Apopka, Florida

The constructed marshes at Lake Apopka, Florida, developed into floating mats (Stenberg *et al.*, 1998). Different vegetation strategies were employed, and all underwent significant conversion to floating mats, over the period 1990–1995. In 1995, 73% of natural succession areas contained floating vegetation mats, while 55% of planted sites were floating.

FIGURE 3.25 A section through the floating mat at the Houghton Lake, Michigan, treatment wetland. The cattail plants have been clipped about 15 cm above the mat, which is about 30 cm thick. Note the black sediment in the rhizome and root matrix.

Belowground biomass declined as roots and rhizomes shifted to floating vegetation mats.

STA1W, Florida

Cell 2 of Stormwater Treatment Area 1W (STA1W) was constructed on peatlands formerly in agriculture. Cattails developed early in the project life, comprising about 50% cover of the 413-ha wetland. Changes in the mode of operation of this cell after six years of operation caused greater water depths, and the cattail areas separated from the base substrate and became floating mats. Because there was less than 100% cover, these floating islands moved with the wind. The water was shallow enough that portions of the mat bottom scraped the base substrate below, creating suspension of the base soils. The water in the cell was therefore very turbid, and the associated nutrients were exported from the wetland. Performance was severely impacted. The cell was later reconfigured, and partially converted to submerged aquatic vegetation.

Floating Mat Constructed Wetlands

Floating mat wetlands have been intentionally employed at many sites, and in great variety. In general, they do not employ SSF substrates, but may employ floating matrices for plant support. In terms of performance, these will later be included in the category of free water surface wetlands, although in fact the mat covers the water surface. The float method has several potential advantages:

- It directly takes up nutrients from water column, and does not remobilize the nutrients in sediment.
- It enables the use of a diversity of aquatic plants.
- It could be used in any water body, regardless of its depth and bottom characteristics.
- Biomass harvesting is theoretically easy.

The principal drawback for raft systems is cost: the frames are expensive. Support structures for the mat are quite varied in design. Several ideas have been patented (Balogh, 1982; Ishikawa and Mizuno, 1988; Hondulas, 1994), and complete units are commercially available (van Duzer, 2004). Here the vegetative character of such constructed mats is described via examples.

New Zealand and Australia

The plant *Glyceria maxima* is capable of being established as a floating mat without the assistance of any support frames. Work at Hamilton, New Zealand, at mesocosm and pilot scale, showed that *Glyceria* mats could be excised in sections from existing treatment wetlands, and floated on the water in a new treatment wetland (Van Oostrom and Russell, 1994).

Wen and Recknagel (2002) implemented polyethylene foam floats, planted with parrot feather (*Myriophyllum aquaticum*), water couch (*Paspalum paspalodes*), and waterbuttons (*Ranunculus repens*). The intent was to treat irrigation drains, fields, or treatment ponds in order to eliminate dissolved phosphorus. Phosphorus removal rates in the range of 0.043–0.086 $g/m^2 \cdot d$ were measured as bioaccumulation in plant tissues.

United Kingdom and Europe

Hiley (1990) reports that "raft lagoon" systems were built at Highroyd, Bishop Wilton, Pattrington, Yorkshire, United Kingdom. These were supported by a buoyant geotextile of 5 cm mesh, and contained a variety of plants for testing purposes. Cattails (*Typha*) worked well, as did sweetflag (*Acorus*), marsh marigold (*Caltha*), and bentgrass (*Agrostis*). However, *Iris, Nuphar,* and *Spartina* were unsuccessful. Small-scale trials indicated that *Phragmites* and *Phalaris* would be good raft candidates.

London's Heathrow airport pilot tested floating rafts of both *Typha* and *Phragmites*, at the scale of 6 × 7 m, operated with detention times of less than one day. Subsequently, a 1.2-ha floating raft wetland was built, and planted with *Phragmites*. The design detention time was just over one day (Revitt *et al.*, 2001; Richter *et al.*, 2003; see Chapter 13).

Artificial floating meadows have been piloted near Budapest, Hungary (Lakatos, 1998). Rectangular wooden frames were filled with plastic netting, and planted with a wide variety of plants. Sixty-seven species were tested, of which 20 either died or did not do well, and 47 grew normally or better. Species found to be suitable included *Alisma, Glyceria, Sagittaria, Sparganium,* and *Typha*. Removals ranged from 40% for phosphorus to 98% for oxidized nitrogen.

The School of Agricultural Engineering of Madrid, Spain, has developed several applications of floating mat systems, which are termed *floating macrophyte filter* (FMF) systems. Systems have been installed at the communities Aviles, Coy, and Doña Inés; at a pig farm in Lorca, Spain (Figure 3.26); as well as at single-family residences. Plants that have been used are *Phragmites*, *Sparganium*, *Schoenoplectus (Scirpus)*, *Iris pseudocorus,* and *Typha*. *Typha* species have shown the best results, with high growth and treatment rates (Curt *et al*., 2005).

(a)

(b)

FIGURE 3.26 Floating macrophytes filter on the pig farm owned by the cooperative Sociedad Cooperativa GAMUR (Ganaderos de Murcia) in Lorca, Spain. (a) Start-up with plants in floating containers. (b) System after partial grow-in. (Photos courtesy Fundación Global Nature: http://www.macrophytes.info/galeria_imagenes.html.)

North America

A variety of floating platform wetlands have been used in the United States, mostly on an experimental, pilot basis. For example, an open water area in a peatland near Madison, Wisconsin, was covered with a matrix of logs, leaf bales, and wire mesh (Hefty, 2002). Planted species included pickerelweed (*Pontederia cordata*), bulrushes (*Scirpus acutus, Scirpus fluviatilis*), burreed (*Sparganium eurycarpum*), and arrowhead (*Sagittaria latifolia*). Muskrats ate the river bulrush, pickerelweed, and arrowhead. Floating rafts were installed in Lake Mead, Nevada, for the purpose of improving water quality (Boutwell, 2001). Cattails (*Typha domingensis*) were found to be more successful than various bulrushes (*Scirpus* spp.).

The removal of metals from mine waters may lead to accumulations in sediments that could be dangerous to sediment-foraging organisms. The use of floating systems allows the accumulation to occur in deep-water locations, as a result of processes in the root zone that drop metal-laden materials. Such systems have been implemented in several locations in Canada, including Buchanans, Newfoundland, Sudbury, Ontario, and Kitimat, British Columbia (Smith and Kalin, 2001). Frames were constructed from timber and snow fencing, buoyed by extruded polystyrene (XPS), and planted with narrow-leaved cattail (*Typha angustifolia*). These systems have been found effective for suspended solids removal.

Woody Plants

Many natural freshwater wetlands contain a variety of woody species. In the southern United States, swamps commonly contain cypress (*Taxodium*), gum (*Nyssa*), and swamp white oak (*Quercus*). In northern North America, species include white cedar (*Thuja*), spruce (*Picea*), red maple (*Acer*), willow (*Salix*), and alder (*Alnus*). However, forested wetlands have only rarely been constructed, and then in tropical or subtropical climates. This is probably more due to the perception of a long grow-in period, rather than to any potential deficiencies in treatment capability. Greenway and Bolton (2001) suggest that this is possibly an oversight:

> Little attention has been given to the use of tree species as candidates for constructed wetlands and yet, woody species may have additional advantages such as higher nutrient uptake, higher rates of primary productivity, higher nutrient storage capacity (biomass sink potential), lower maintenance (due to greater tree longevity) and the production of useful resources. Harvesting biomass for resources also removes the accumulated nutrients, which could be recycled through mulch.

At present, the applications of forested constructed wetlands are in the following principal categories:

- Melaleuca (tea tree) systems
- Mangrove systems
- Willow systems
- Forested edges or bands in stormwater wetlands

The data from constructed woody wetlands is too sparse to analyze with the same degree of thoroughness as emergent marshes. The beginnings of a performance basis are discussed here.

Melaleuca

The genus *Melaleuca* is found in tropical and subtropical climates. It is comprised of 250 species, and was formerly prolific in the lowlands of Australia. It is a modern-day widespread nuisance invader in Florida. It can form a dense stand that crowds out virtually all other species. It is also known as "paperbark" because of its soft, paper-like bark, and some varieties are economically valuable because of the oils that they produce. It has been shown that these trees may be fostered in constructed wetlands, and provide excellent treatment capability (Bolton and Greenway, 1997; Bolton and Greenway, 1999a; 1999b; Greenway and Bolton, 2001). *Melaleuca alterniflora* grew rapidly, and accumulated about 5,000 gdw/m^2 in a 21-month irrigation period with secondary effluent (Bolton and Greenway, 1997). The trees may be harvested, with regrowth occurring if a dry-down period is included.

Mangroves

Mangroves are one of the few woody species that can tolerate saltwater environments. There are many species, including *Kandelia candel*, *Avicennia marina*, and *Rhizophor*a spp. in Asia; and *Avicennia germinans* and *Rhizophora mangel* in tropical South America. Integrated mangrove-aquaculture systems are currently practiced throughout Asia, including the traditional *gei wai* in Hong Kong and *tambak* in Indonesia (Primavera, 2000). Shrimp pond effluents are treated in pond-mangrove systems, most often involving natural mangrove stands. This practice has been criticized as causing degradation of the mangroves (Gautier, 2002). The use of natural mangrove stands has not been particularly effective in Colombia (Gautier *et al.*, 2001). Natural mangrove stands were studied at Shenzhen, China, for the purpose of treating settled sewage, and survived a moderately heavy loading (Wong *et al.*, 1997). Constructed (transplanted soils and plants) mangrove wetland mesocosms were effective in controlling metals in applied sewage (Tam and Wong, 1997). Boonsong *et al.* (2003) studied a 1.5-ha wetland planted with four species of mangroves, for the treatment of dilute, settled sewage. Good removals were found for many common wastewater parameters, at three and seven days' detention, over a year-long study. The authors cautioned that the effects on the plants remained to be assessed.

Willows

Willows (*Salix* spp.) grow rapidly and are water-loving plants. An entire subset of treatment wetland technology has developed, in which the transpiration capabilities of willows is used to create zero-discharge wastewater treatment systems, primarily for single residences and small communities (Gregersen and Brix, 2001; Brix and Gregersen, 2002; Brix and Arias, 2005). Many willow systems are now functioning in Denmark (Figure 3.27).

Principal features are zero discharge and recycle of part of the nutrients via harvested willow biomass. Danish guidelines have been published (Gregersen *et al.*, 2003). Willow facilities generally consist of 1.5-m-deep high-density polyethylene-lined basins filled with soil and planted with clones of willow (*Salix viminalis* L.). The surface area of the systems depends on the quantity and quality of sewage to be treated and the local annual rainfall. For a single household in Denmark, the area needed typically is between 120 and 300 m^2. Settled sewage is dispersed underground into the bed under

FIGURE 3.27 A willow treatment system at Pileanlag, Denmark. (Photo courtesy C. Arias.)

pressure. The stems of the willows are harvested on a regular basis to stimulate the growth of the willows and to remove some nutrients and heavy metals.

The total annual water loss from the systems is assumed to be 2.5 times the potential evapotranspiration at the location as determined by climatic parameters, and is partially compensated by precipitation. In small systems, the vegetation experiences enhanced evaporation from the "oasis effect," resulting from warmer and dry air flowing across an area of plants. In addition, there is also the "clothes-line effect," where the vegetation height is greater than that of the surroundings and may increase evaporative loss. Therefore, evapotranspiration from isolated expanses, on a per unit area basis, may be significantly greater than the calculated potential evapotranspiration.

One third or one half of the willows are harvested every year to keep the willows in a young and healthy state with high transpiration rates.

Woody Plants in Stormwater Wetlands

Wetland-tolerant trees have been used as an internal landscaping feature in some urban stormwater wetlands. For instance, cypress (*Taxodium*) was used in the Greenwood urban wetland in Orlando, Florida, and red maple (*Acer*) in the Tollgate urban wetland in Lansing, Michigan.

Wastewater and Natural Forested Wetlands

Natural wetlands have been used for wastewater treatment in modern times, including forested systems. These wetlands are to some degree engineered to accommodate and treat incoming wastewater. In some cases, forested wetlands have accommodated wastewater with only small effects on the preexisting ecology, while in other, more numerous cases, the original ecology has been severely disrupted. In many cases, woody species have not survived, including both shrubs and trees. Replacement communities are often soft tissue herbaceous plants, notably *Typha* spp. Forested northern bogs have not provided good long-term treatment, and have been markedly altered by wastewater additions (Guntenspergen *et al.*, 1980; Nichols and Higgins, 2000).

The water quality performance of forested wetlands can differ markedly from that of emergent marshes, submerged vegetation, and floating plant systems. For instance, Kadlec and Knight (1996) reported that the phosphorus removal rate constant for 63 emergent marsh wetlands averaged 13.1 m/yr, while for 11 natural forested wetlands, it averaged 3.1 m/yr. In contrast, the removal of nitrate in forested wetlands is quite good, averaging 96 ± 5% for five systems in the southeastern United States (Boustany *et al.*, 1997; NADB database, 1998).

This book does not focus on such systems, but the reader may wish to consult the appropriate literature if faced with the need to evaluate them. Accordingly, a few brief examples from the available literature are provided here.

Northern North America

Forested wetlands abound in Minnesota, Michigan, and the Canadian provinces, and some have been the recipients of treated wastewater. Monitoring has been conducted at several sites. Based upon results, few if any new projects have been permitted in the past three decades.

Kinross, Michigan

Prior to 1977, the Kincheloe Air Force Base was in full operation with an estimated population of 7,500 people. The plant treated an average wastewater flow of 2,300 m^3/d with primary clarifiers, trickling filters, and secondary clarifiers, followed by gravity discharge to the adjacent 300-ha wetland (Kadlec and Bevis, 1990). Approximately one third of this area was impacted by the discharge. Nitrogen removal was effective in this wetland, with about 99% reduction in both ammonia and oxidized nitrogen. BOD_5 and TSS were reduced 64% and 94%, respectively. Phosphorus removal was complicated by the time sequence of wastewaters and treatment. The base used very large quantities of phosphate detergents to wash airplanes, and the original wastewater treatment plant (WWTP) did not remove phosphorus. An unknown but large phosphorus load was delivered to the wetland and presumably partially trapped. Subsequently, the base closed, and the WWTP added phosphorus removal. Therefore, low-phosphorus water entered the wetland, which released considerable phosphorus and caused outflows higher than inflows.

It is probable that this peatland was at one time a shallow lake basin that had filled and developed into a palustrine acid/peat wetland system typical of the region. The original wetland probably contained sedges (*Carex* spp.), leatherleaf (*Chamaedaphne calyculata*), sphagnum (*Sphagnum* spp.), and sparse black spruce (*Picea mariana*), based on remnants and adjacent ecosystems. The wastewater promoted a shift to a monoculture of cattails (*Typha latifolia*). Peat in the near discharge area largely disappeared, leaving a floating mat of cattails over ooze. A trend to a cattail monoculture occurred, and the vicinity of the discharge now consists of a cattail monoculture surrounded by upland forest. Small, isolated duckweed ponds dot the cattail stand. Occasional remnant living and dead black spruce "islands" are present. Aerial photos from 1939, prior to wastewater addition, show no evidence of cattail (Kadlec and Bevis, 1990). The gradual expansion of the *Typha* monoculture from the discharge point downgradient continued up to 1981, at which time the entire watercourse had converted to cattail. Figure 3.28 illustrates this change-over process for another system at Biwabik, Minnesota.

Bellaire, Michigan

The wetland is peat-based, and vegetated by white cedar (*Thuja occidentalis*), spruce (*Picea* spp.), and black ash (*Fraxinus nigra*), grading to a sedge-shrub community containing *Carex* spp., alder (*Alnus* spp.), red osier dogwood (*Cornus*

FIGURE 3.28 Cattails (foreground) intruding on a spruce forest at Biwabik, Minnesota, after 15 years of effluent addition. The lagoon discharge is in the extreme foreground, and water flows toward the forest in the background.

stolonifera), and poison sumac (*Rhus vernix*; Kadlec, 1983). The wetland receives groundwater discharge from the adjacent sand hills. In the early 1970s, a sewage collection system was put in place, bringing the wastewater to a double-facultative lagoon treatment facility. Wastewater was discharged from the lagoons, in spring and again in fall into forested lake margin wetlands. These intense, high volume discharges flushed debris from the wetlands to the lake, and passed very quickly without substantial treatment. In 1976, the discharge was altered to occur throughout the entire unfrozen season, and was spread more uniformly across the wetland. In 1982, the discharge was moved to secondary locations. In 1989, the treatment process was upgraded to include sand/carbon filtration and alum addition, and the discharge was moved to a third set of locations.

Early in the project life, both nitrogen and phosphorus were reduced to low values. However, phosphorus removal became ineffective after several years (Kadlec, 1983). During the recovery period 1982–1987, the wetland then received only the groundwater discharge. The wetland removed 1,097 kg of phosphorus over the six-year period of controlled water discharge (1.02 $g/m^2 \cdot yr$), resulting from a decrease from 2.71 to 0.29 mg/L. In the six years following termination of discharge, the wetland released 64 kg of phosphorus, or 6% of the amount removed.

The trees in the first irrigation zone were killed, probably by a combination of long hydroperiod and root zone erosion. Black ash was the most resistant to destruction. The replacement community was comprised of soft-tissue plants, dominated by *Typha* and jewelweed (*Impatiens capensis*). The second irrigation area also suffered tree destruction (Figure 3.29). The third irrigation area had distribution of water in grassed strips in the forest, and to date, the forest has survived well in that area (1989–present).

FIGURE 3.29 Tree death at the Bellaire wetland. All of the trees in the foreground, save one, are standing or fallen dead in this July photo. They have been replaced by soft-tissue plants after ten years of wastewater addition.

Houghton Lake, Michigan

This system is the oldest continuously monitored natural wetland treatment project in the United States (U.S. EPA, 1993f; Kadlec and Knight, 1996). Wastewater from this residential community is collected and transported to two 2-ha aerated lagoons, and is then stored in a 12-ha pond for summer disposal to a 600-ha peatland locally known as the Porter Ranch Peatland. The Porter Ranch Peatland has received better-than-secondary wastewater for 30 years, since 1978, during which hydrology, water chemistry, soils, and vegetation were studied. Hydraulic loadings to the impacted area are less than 1 cm/d, and occur only in the unfrozen season. Water quality has been consistently improved by passage through the wetland. Nitrogen and phosphorus are removed to background. Phosphorus removal was found to first order with a rate constant of 11 m/yr. Five years were required to stabilize nutrient fronts (Kadlec, 1997a).

The ecosystem has changed character markedly in the zone of discharge (Kadlec, 1993). Water regime changed to a greater duration of flooding. Major community changes took place within the irrigation area. Vegetation species composition shifted from sedges (*Carex* spp.) and shrub willows (*Salix* spp.) to cattail (*Typha latifolia* and *Typha angustifolia*) and duckweed (*Lemna minor*; Figure 3.30). About 80 ha of wetland have been altered. Aboveground biomass tripled, and several centimeters of soil accreted. Some plant species—all woody species and sedge—were totally lost in the discharge area. Cattail and duckweed have totally taken over, and are increasing in abundance and range. An aspen community near the pipeline completely succumbed in 1983. A second aspen island located 500 m downgradient had also totally succumbed by 1984. The aspen on the edges of the peatland have died in backgradient and side locations where the shore slopes gradually. The alteration of the water regime was the probable cause of tree death along much of the wetland perimeter, in a band up to 50 m wide at a few locations. Long-dead timber at these locations indicates that similar events may have occurred naturally in the past.

Southern United States

Pioneering research on wastewater irrigation to cypress domes was conducted by the University of Florida in the 1970s (Ewel and Odum, 1984). Since that time, and based in major part upon that research, guidelines and regulations have been developed at both the state and national level (see, for instance, U.S. EPA, 1985a; Schwartz, 1989).

North Myrtle Beach, South Carolina

Pretreated wastewater from aerated lagoons is distributed to 69-ha Bear Bay, a natural forested wetland, through a series of gated aluminum pipes supported on wooden boardwalks (U.S. EPA, 1993f). The vegetation of Bear Bay is dominated by tree species, including sweet gum (*Liquidambar styraciflua*), red maple (*Acer rubrum*), pond pine (*Pinus serotina*), and loblolly pine (*Pinus taeda*). Operation of the full-scale system began in October 1990. On the basis of an estimated area of 28 ha, treated wastewater flows are equal to annual average hydraulic loading rates between 0.2 and 0.6 cm/d. The water quality entering the wetland was partially nitrified secondary. The five-year average mass removal efficiencies in Bear Bay were at least 88% for BOD_5, TSS, NH_4-N, TN, TP, and UOD (Kadlec and Knight, 1996).

Decreases in tree canopy density were observed in areas continuously flooded with wastewater. This allowed expansion of herbaceous communities dominated by pennywort (*Hydrocotyle* spp.) and duckweed (*Lemna* spp.) (Kadlec and Knight, 1996). This phenomenon was visually evident near the treated wastewater discharge area due to early and prolonged leaf fall and mortality of susceptible tree species including loblolly pine, sweetgum, American elm, red maple, and water oak.

Reedy Creek, Florida

The 35-ha Reedy Creek natural wetland was vegetated with water ash (*Fraxinus caroliniana*), magnolia (*Magnolia virginiana*), and blackgum (*Nyssa sylvatica*), and received 12,000 m^3/d of better-than-secondary pretreated wastewater (Knight *et al.*, 1987; Kadlec and Knight, 1996). Removal efficiencies were 60–80% for BOD_5, TSS, and TN, but less than zero for TP, over a 12-year period of record, 1978–1989. There was not before-and-after sampling of the biological communities. However, in spite of about ten years of wastewater discharges, the wetland still supported a diverse and robust forested wetland plant community in 1988. Stem density and basal area were both high at 3,785 stems per hectare and 38.29 m^2/ha, respectively, and are typical of mature southern coastal plain swamps.

Poinciana, Florida

The 47-ha Boot Wetland in Poinciana, Florida, was a drained and degraded forested wetland, dominated by pond cypress (*Taxodium ascendens*) and blackgum (*Nyssa sylvatica;* Martin *et al.*, 2001). In 1984 it became the recipient of 1,060 m^3/d of nitrified secondarily treated wastewater, which continues to the present. This is an exceedingly low hydraulic loading, amounting to less than rainfall on the wetland, and half the inflow was lost, presumably to infiltration and evapotranspiration. Concentration reductions were good for TP (69%) and TN (48%), the latter reflecting complete removal of oxidized nitrogen. Incoming BOD_5 (2.5 mg/L) and TSS (5.5 mg/L) were so low that there were small increases, presumably to wetland background.

This wetland was in a degraded condition prior to wastewater addition due to forestry, drainage canals, and surrounding land development. Peat oxidation had occurred, and trees were toppling. Compared to the antecedent condition, the structure and function of the system was significantly improved by wastewater irrigation (Martin *et al.*, 2001). The dominance and density of trees was increased. However, the creation of a 100% hydroperiod, and a continuous water depth of 70–90 cm, resulted in water surface

(a)

(b)

FIGURE 3.30 Sedge meadow replacement at the Houghton Lake, Michigan, wetland. All of the sedge (a) disappeared, and was replaced by cattail (b) after wastewater addition.

cover of 100% of floating, leaved plants, including duckweed (*Lemna* spp.), frog's bit (*Limnobium spongia*), and water fern (*Salvinia rotundifolia*). Natural wetlands of the region had a much lower frequency of inundation, and consequently fostered a different understory community type, dominated by water hyssop (*Bacopa carolinianum*). Thus a healthy wetland resulted, but one with an unnatural vegetative structure.

Thibodaux, Louisiana

The Thibodaux, Lousiana, site consists of an almost permanently flooded, subsiding, forested wetland, containing cypress (*Taxodium distichum*) and gum (*Nyssa aquatica*). Since 1992, the 231-ha wetland has received secondarily treated municipal wastewater at the average rate of 15,140 m^3/d. Loading amounts to about 0.27 cm/d of water, and 124 kgN/ha·yr (Boustany *et al.*, 1997). The receiving wetland had been hydrologically altered by some combination of levees, spoil banks, highways, oil and gas access roads, or railroad lines (Day *et al.*, 1999). At such low loadings, effects on the ecosystem structure are believed to be absent or at least long-delayed.

From 1992 through 1996, the mean annual reduction oxidized nitrogen, the dominant form of nitrogen in the effluent, ranged from 96% to 99%. From 1992 through 1994, the mean annual reduction of total phosphorus in the wetland ranged from 33% to 71%. High rates of accretion and burial of sediments in the subsiding system provides a permanent sink for phosphorus (Zhang, 1995). Results from several ongoing and completed studies of wastewater treatment in other forested wetlands of the region indicate that they are achieving the ecological goals of enhancing effluent water quality, stimulating vertical accretion, and increasing productivity (Day *et al.*, 2004). Economically, the savings are substantial for small communities and nontoxic industrial processors (Breaux *et al.*, 1995).

Emergent Soft Plants

By far the largest number of treatment wetlands utilize soft tissue plants (herbaceous vegetation), as discussed in the remainder of this chapter. Emergent vegetation is the most common choice; because these plants fit a wide variety of niches in wetland ecosystems, planting stock is often available through commercial plant nurseries, and they spread through lateral rhizomes, which allows the relatively rapid development of an emergent plant canopy.

Surface Flow Wetlands

Emergent wetland plants provide a wide range of treatment mechanisms in FWS wetlands, (Sinclair Knight Merz, 2000) including:

- Increased sedimentation by reducing wind-induced mixing and resuspension
- Additional surface area in the water column, which increases biofilm biomass and soluble pollutant uptake
- Increased surface area for particle interception
- Shade from the plant canopy over the water column to reduce algae growth
- Induced flocculation of smaller colloidal particles into larger, settleable particles

Most of these mechanisms are structural in nature. Consequently, selecting the "perfect" species is not nearly as important as establishing a functional plant canopy. As microbiological transformations within the wetland are a function of area available for biofilm growth, the creation of surface area by emergent aquatic plants and associated leaf litter is an important contribution to the treatment process. Plant species that provide structure year-round generally perform better than species that die below the water line after the onset of cold temperatures. For these reasons, fast-growing emergent species that have high lignin contents and that are adapted to variable water depths are the best suited for FWS systems. Wetland plant genera that most successfully meet these criteria include cattails (*Typha* spp.), bulrushes (*Scirpus* spp.), and common reed (*Phragmites australis*).

Vegetation types in FWS wetlands exhibit small performance differences, but these differences are often masked by other unavoidable differences in comparable wetlands. At the time of this writing, the case for superiority of a particular plant species has not been proven or disproven. The evidence points toward minimal net differences among plant species. A more diverse mix of plant species will be better able to accommodate changes in water quality and flow. In other words, a polyculture is preferable to a monoculture.

Table 3.10 shows plant species used for initial planting of FWS wetlands listed in the NADB v.2 (NADB database, 1998). That database contains only an early subset of FWS systems, and does not include many recently built systems.

TABLE 3.10
Species Reported as the First Dominant in Constructed FWS Wetlands in the North American Database, Version 2.0

Scientific Name	Common Name	Number of Wetlands
Typha spp.	Cattail	206
Scirpus spp.	Bulrush	49
Juncus spp.	Rush	19
Phalaris arundinacea	Reed canary grass	15
Phragmites australis	Common reed	13

Source: Data from NADB database (1998) *North American Treatment Wetland Database* (*NADB*), *Version 2.0*. Compiled by CH2M Hill, Gainesville, Florida.

Subsurface Flow Wetlands

Compared to FWS wetlands, subsurface (SSF) systems are much less dependent on plants to sustain their treatment processes. A SSF wetland will require planting because nearby seed banks are typically lacking and the gravel media is not optimal for seed germination.

Small performance differences among vegetation types also exist for SSF wetlands (Brisson *et al.*, 2006), but since the role of plants is small in these systems, plant effects are masked by other unavoidable differences in comparable wetlands. Therefore, no conclusive results could be found in a review of 47 species studied in 27 different comparative investigations (Brisson *et al.*, 2006). Speculatively, plants that have significant root penetration into the bed media are likely to enhance treatment. Effects of plant root systems include:

- Additional surface for biofilm created by the root system.
- Oxygen diffusion from root surfaces into the water column. (However, this plant-mediated oxygen transfer is very small relative to the applied internal and external organic loadings in most SSF systems.)
- Chemical exudates used by the plants detoxify the root environment.
- Additional fungi species introduced by the plants.
- Symbiotic bacteria introduced by the plant root systems.

The combined effect of these phenomena is a larger and more diverse microbial community within the SSF bed. Comparing the results of plant investigations in different SSF wetlands does not provide compelling evidence that any particular plant species offers superior treatment (Gersberg *et al.*, 1984; DeBusk *et al.*, 1989; Van Oostrom and Cooper, 1990; Batchelor *et al.*, 1990; Knight, 1993).

Designers typically focus on plants that are easy to propagate and are able to survive in the relatively hostile environment of a SSF wetland. The most frequently used plant species worldwide is *Phragmites australis* (common reed). This species has remarkable growth rates, root development, and tolerance to saturated soil conditions. Common reed is also known to provide some ancillary benefits in terms of wildlife habitat in the United Kingdom (Merritt, 1994).

Phragmites has a long history of cultural use and is used almost exclusively for wetland treatment systems in Europe. However, in many areas of the United States, *Phragmites* is considered an exotic and invasive plant species, and therefore use of this species in North America has been limited. Table 3.11 shows plant species used for initial planting of SSF wetlands listed in the NADB v.2 (NADB database, 1998). That database contains only a small and early subset of SSF systems. It does not include the current large numbers of small systems (Wallace and Knight, 2006).

Examples of Modern Emergent Community Choices

Given the apparent lack of performance differentials, many recent wetlands have been planted with a view to biodiversity and aesthetics. In the United States, decorative plants such as canna lilies (*Canna flacida*) and yellow iris (*Iris pseudocorus*) have historically been used at a number of HSSF sites, including Denham Springs and Carville, Louisiana (Wolverton, 1989). Although it is recognized that volunteer vegetation will continue to alter these systems, they have been established as communities with a variety of compatible species. Numerous systems have been so initiated; a few examples are given here.

FWS Systems

Wakodahatchee, Florida

The Wakodahatchee Wetlands were created by converting a series of percolation ponds into flow through marshes (Bays *et al.*, 2000; Hobbs *et al.*, 2003). The wetland area totals 16 ha of wetted acreage, with individual wetland cells ranging from 1 to 5 ha. Deep zones, consisting of ponds 2 m in depth and variable in width and oriented transverse to the direction of flow, are interspersed throughout each wetland to maximize hydraulic retention time, create aquatic habitat, and equilibrate flow. Marshes comprise about 70% of the wetland area, vegetated by native emergent, forested, and transitional wetland species designed to emulate native South Florida wetland plant communities. Emergent marsh zones are composed of bulrush (*Schoenoplectus* (*Scirpus*) *validus*, *S. californicus*), duck-potato (*Sagittaria lancifolia*), arrowhead (*Sagittaria latifolia*), spikerush (*Eleocharis cellulosa*), fireflag (*Thalia geniculata*), and pickerelweed (*Pontederia cordata*). Herbaceous species planted at the upper edge of the marsh zone include sawgrass (*Cladium jamaicense*), Fakhahatchee grass (*Tripsacum dactyloides*), and Gulf muhlygrass (*Muhlenbergia capillaris*). Forested species also planted at the marsh edge include cypress (*Taxodium ascendens*), pond apple (*Annona glabra*), Carolina willow (*Salix caroliniana*), red maple (*Acer rubrum*), and buttonbush (*Cephalanthus occidentalis*). Duckweed (*Lemna* spp.) has become naturally ubiquitous throughout the wetland, but with significant seasonal fluctuations in total cover. Treatment of the secondary influent is comparable to other systems in the region.

TABLE 3.11
Typical Species Used in Constructed SSF Wetlands in the North American Database, Version 2.0

Scientific Name	Common Name	Number of Wetlands
Typha spp.	Cattail	5
Scirpus spp.	Bulrush	55
Phalaris arundinacea	Reed canary grass	1
Phragmites australis	Common reed	9

Source: Data from NADB database (1998) *North American Treatment Wetland Database* (*NADB*), *Version 2.0*. Compiled by CH2M Hill, Gainesville, Florida.

Victoria, Texas

Wastewater from a chemical plant is treated in a train that includes a 21-ha FWS wetland (Reitberger *et al.*, 2000). The vegetation selected for wetland establishment was a varied mixture. Shallow zones included bulrushes, *Schoenoplectus* (*Scirpus*) (*actus, validus, californicus, americanus, pungens*), plus arrowhead (*Sagittaria graminea*), giant cutgrass (*Zizaniopsis miliacea*), rushes (*Juncus effusus*), spikerush (*Eleocharis* spp.), and sedges (*Carex* spp.). Deeper zones were planted with coontail (*Ceratophyllum demersum*), pondweed (*Potamogeton* spp.), pickerelweed (*Pontederia cordata*), and water lilies (*Nymphaea elegans*). The wastewater treatment process is working well, with COD removal above 99%, and complete removal of nitrate and nitrite, since start-up (Beeman and Reitberger, 2003). The wetland has exceeded expectations for polishing, buffering, and community value.

Lapeer, Michigan

The domestic wastewater from a small manufacturing facility is pretreated in a septic tank, and then discharged to a pond followed by a FWS wetland. The pond had addition of yellow water lilies (*Nuphar lutea*), and developed a fringe of cattail (*Typha* spp.). The wetland was vegetated with hardstem and softstem bulrush (*Scheoenoplectus* (*Scirpus*) *acutus* and *Scheonoplectus* (*Scirpus*) *validus*), together with sedge (*Carex lacustris*), arrowhead (*Sagittaria latifolia*), water plantain (*Alisma plantago-aquatica*), pickerel weed (*Pontederia cordata*), and giant burreed (*Sparganium eurycarpum*). Cattail was an immediate invader in the wetland. Water quality data were within the expected range.

SSF Systems

Jackson Meadow, Marine on St. Croix, Minnesota

This conservation development uses two HSSF constructed wetlands to treat wastewater from a 64-home residential subdivision. Each 0.1-ha wetland cell is designed to treat up to

FIGURE 3.31 Vegetative community of a HSSF wetland at Jackson Meadow, Marine on St. Croix, Minnesota. This system features a vegetative community dominated by river bulrush (*Schoenoplectus* (*Scirpus*) *fluviatilis*), an unplanted colonizer—reed canary grass (*Phalaris arundinacea*)—and New England aster (*Aster novae-angliae*).

34 m^3/d of domestic wastewater. Primary treatment is provided by septic tanks, and wetland effluent is infiltrated back into the soil for recharge of the surficial aquifer. Both systems have consistently met permit limits established by the Minnesota Pollution Control Agency. To protect against freezing in the cold Minnesota winters (temperatures below –30°C are possible), the wetland cells are insulated with a layer of peat mulch 15 cm thick (Wallace and Nivala, 2005). The presence of the peat mulch creates an unsaturated rooting zone that shifts the competitive advantage way from obligate wetland plants towards facultative wetland plants (U.S. Army Corps of Engineers, 1987).

The wetland cells were planted with a variety of native plants. Although the two systems are only 0.8 km apart, they were established in different years (1999 and 2002), and development of the vegetative communities has proceeded along different lines. The north treatment system features a vegetative community dominated by river bulrush (*Schoenoplectus* (*Scirpus*) *fluviatilis*), an unplanted colonizer (reed canary grass [*Phalaris arundinacea*]), and New England aster (*Aster novae-angliae*; Figure 3.31). The south treatment system features a vegetative community dominated by woolgrass (*Scheonoplectus* (*Scirpus*) *cyperinus*), giant burreed (*Sparganium eurycarpum)*, arrowhead (*Sagittaria latifolia*), and prairie cordgrass (*Spartina pectinata*). Cattail (*Typha angustifolia*) was planted in both systems but did not persist beyond the first growing season.

Lutsen Resort, Lutsen, Minnesota

This lodging company operates a variety of resort properties along the North Shore of Lake Superior. Two HSSF and two VF systems have been built over the period from 1997 to 2005; the systems are small (less than 400 m^2), and design flows range from 10 to 80 m^3/d. All systems consistently meet their permit limits. The systems are insulated with a layer of peat to prevent freezing in the cold Minnesota winter. A variety of native wetland plants were used in the systems. Early projects utilized planting of dormant rhizomes in the fall. This was generally unsuccessful due to grazing pressure from whitetail deer (*Odocoileus virginianus*) over the winter months (Wallace *et al.*, 2000).

In the first system (a HSSF wetland planted in 1997) the plant community has evolved over time to include cattail (*Typha angustifolia*) and bulrush (*Schoenoplectus* (*Scirpus*) *fluviatilis*); prairie cordgrass (*Spartina pectinata*) and iris (*Iris pseudocorus*) have also become dominant (Figure 3.32).

FIGURE 3.32 Vegetative community of a HSSF wetland at Lutsen, Minnesota. Plants in the inlet zone are dominated by arrowhead (*Sagittaria latifolia*), with cattail (*Typha angustfolia*) in the background. The vegetative community is approximately eight years old at the time of this photo.

Undesirable plant species have invaded the system over time (see the subsequent section on "weeds"), including willows (*Salix* spp.) and Canada thistle (*Cirsium arvense*), necessitating occasional management control (see Part II).

The second wetland (a HSSF system planted in 1998) has been dominated by woolgrass (*Schoenoplectus* (*Scirpus*) *cyperinus*) and green bulrush (*Schoenoplectus* (*Scirpus*) *atrovirens*). The two VF wetlands (planted in 2004 and 2005) still have juvenile plant communities dominated by biennials, such as black-eyed susan (*Rudbeckia hirta*). Due to their unsaturated flow conditions, it is likely that the VF wetlands will develop plant communities very different than the saturated flow HSSF wetlands, despite the fact that all systems are located in the same climatic conditions within 8 km of one another.

3.5 WEEDS

A "weed" in a constructed wetland is a plant that has not been intentionally planted and possesses one or more characteristics viewed as undesirable. Many species can quickly invade and colonize new treatment wetlands, and may be regarded as weeds, depending upon local opinion. As varied opinions exist around the world, it is not possible to generalize an overall list of "weed" species. For instance, in the Czech Republic, common reed (*Phragmites australis*) and reed canary grass (*Phalaris arundinacea*) are often used for SSF wetland systems because they are native to the country (Vymazal, 1998). In the United States, resource managers often discourage the use of these plants because they are considered as nonnative in most regions of the United States. In addition, these plants are very aggressive, and considerable effort is required to remove them from SSF systems.

The question is whether or not the weeds deteriorate the treatment efficiency of the system. It seems that in most cases where weeds occur or took over the originally planted species, the treatment effect is not hampered. The problem with herbaceous weeds is the aesthetics and, especially in the United States, the occurrence of unwanted not-native species. The problem with woody weeds could be more serious especially in subsurface systems that are commonly lined with plastic liners. In this case there is a danger of root penetration through the liner and subsequent water leakage.

However, in well established stands the weedy species are usually limited to the wetland margins because plants typically used in constructed wetlands are quite robust, and it is difficult to outcompete these plants once they are established. Regardless of the location of the wetland, there is always the potential that some type of "undesirable" plant will introduce itself, especially those spreading easily by seeds. Consequently, any operations and maintenance plan for a wetland treatment system should address removal and management of undesirable plant species. Typical plant species invading constructed wetlands include purple loosestrife (*Lythrum salicaria*), cottonwood (*Populus deltoides*), willow (*Salix* spp.), stinging nettle (*Urtica diocia*), and in North America, common reed (*Phragmites australis*).

Examples of Weeds in Treatment Wetlands

The submersed macrophyte hydrilla (*Hydrilla verticillata*) is native to the warm areas of Asia. It was first discovered in the United States in 1960 (Langeland, 1996). This plant is well adapted to life in submersed freshwater environments, and has spread rapidly through portions of the United States, where it has become a serious weed. Where the plant occurs, it displaces native aquatic plant communities, and adversely impacts freshwater habitats by forming a dense surficial cover. *Hydrilla* has invaded hundreds of hectares of the Florida stormwater treatment areas (STAs). Its performance for water quality improvement is somewhat poorer than other SAV species (DB Environmental, unpublished results, 2006).

Phragmites australis is a widely distributed clonal grass species, ranging all over Europe, Asia, Africa, America, and Australia. Extensive reed beds are protected in Europe because of their important ecological functions. In contrast, the rapid expansion of *P. australis* in North America, particularly along the Atlantic coast, is considered a threat to biodiversity. Although *P. australis* was a component of marshes in New England several thousand years ago, genetic evidence (Saltonstall, 2002; Blossey *et al.*, 2002) has now confirmed that a more aggressive genotype has been introduced to North America, probably in the late 1800s. Dense *Phragmites* stands in North America have decreased native biodiversity and quality of wetland habitat, particularly for migrating waders and waterfowl species. The closest related species is *Arundo donax*, also an invasive introduced species.

Purple loosestrife (*Lythrum salicaria*) is a herbaceous perennial of Eurasian origin that became established in northeastern North America in the early 1800s (Thompson *et al.*, 1987). By the late 1800s it had spread throughout the northeastern United States and southeastern Canada, reaching as far north and west as Manitoba. Since then, it has steadily expanded its local distribution and now poses a serious threat to native emergent vegetation in shallowwater marshes throughout northern North America. Thompson *et al.* (1987) observe:

> It is no small irony that after 50 years of struggle to find some means of breaking up monotypic stands of cattails (*Typha* spp.) to increase wildlife diversity and abundance, wetland managers must now cope with a foreign species that replaces cattail, but unfortunately creates another monospecific community of greatly diminished wildlife value.

The impact of purple loosestrife on native vegetation in North America is disastrous, with almost the entire biomass of some wetland communities displaced. Monospecific blocks of this weed have survived for at least 20 years. Impacts on wildlife indicate serious reductions in waterfowl and aquatic furbearer productivity. Several declining species of vertebrates are threatened with further degradation of their breeding habitats with the continued expansion of purple loosestrife.

Woody plants, especially willows, are opportunistic invaders in some constructed treatment wetlands (Figure 3.33).

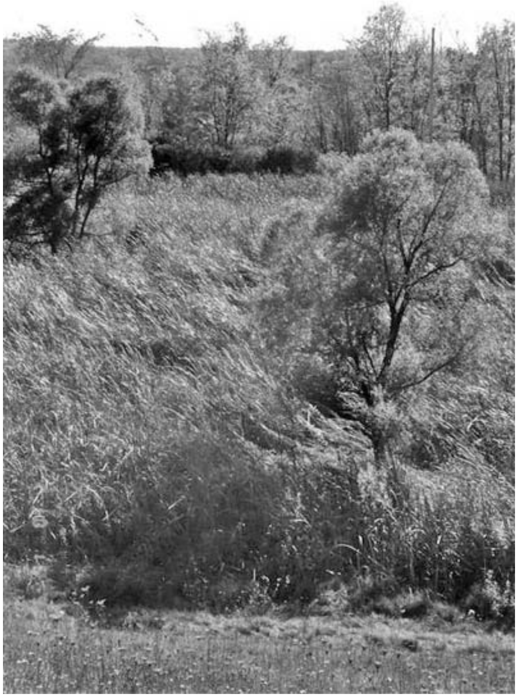


FIGURE 3.33 (A color version of this figure follows page 550) Trees growing in the Vermontville, Michigan, constructed treatment wetland after 15 years of operation.

Although there are no documented failures of wetlands due to tree growth, there is a perception that potential problems could occur. For instance, tree roots can compromise the integrity of containment berms if the berms are of small cross-section. There is also the possibility that tree roots could puncture wetland liners, either plastic membranes or clay layers, and allow wastewaters to seep to groundwater. Another possibility is that large trees would be susceptible to falling over during high wind events if the wetland liner prevents normal propogation of the root systems of the tree.

In contrast to the idea of invasive takeover and destruction of biodiversity, there is the sometimes-held concept that wetlands should contain only the plants selected in design. The view of the Constructed Wetland Association (CWA) of the United Kingdom is that reed beds should be *Phragmites* monocultures as designed. A study of 255 reed bed sites built by Severn Trent Water found that "weed infestation" was a problem at 130 of the sites (51%), defined as more than 25% cover (Cooper *et al.*, 2006a). The perception of CWA is that *Phragmites* is necessary to alleviate clogging of the bed, by several mechanisms including "windrock." Other plants are believed to be capable of functioning only for some weeks or months.

SUMMARY

Wetland environments support a wide variety of bacteria, fungi, algae, and macrophytes (submerged, floating, and emergent). Treatment wetlands have been implemented that use periphyton, algae, submerged macrophytes, floating vegetation, and woody plants, although emergent macrophytes remain the most common choice. In order to survive in a flooded environment, emergent macrophytes transport oxygen from their leaves through their stalks to the root tissues. The majority of this oxygen is used for plant respiration, although some is used to detoxify the rooting environment. For wetlands treating primary effluents, such as many HSSF wetlands, the amount of oxygen that passes into the water column from the plant roots is negligible compared to the wastewater loading, and majority of the root biomass is in the top 20 cm of the wetland bed.

The growth, death, and decay of plant biomass is an important biogeochemical cycle in treatment wetlands and imposes a seasonal cycle on many internal processes. During the growing season, nutrients such as nitrogen and phosphorus are taken up by the plants, and temporarily stored in the plant canopy. This uptake is significant for juvenile ecosystems where the plant canopy being established, and for periods of peak plant growth. At the end of the growing season, nutrients are returned to the system after the emergent portion of the plants die back. The decay of plant biomass imposes nonzero background concentrations for many constituents in treatment wetlands and is important in some treatment processes, such as denitrification. Some portion of the phytomass is resistant to degradation, leading to a net accretion of refractory organic matter in treatment wetlands.

A wide variety of plant species have been used in treatment wetlands, and initial plant selection is a function of hydropattern, climate, and cultural choices. Regardless of the initial planting, the plant community will self-organize over time as additional plant species invade the system. If the project goal is to maintain a specific plant community, human intervention will be required to remove plants that are viewed as being undesirable.

4 Energy Flows

Water temperatures in treatment wetlands are driven by energy flows (gains and losses) that act on the system. During warm conditions, the largest energy gain is solar radiation, and the largest energy loss is evapotranspiration. Energy flows are cyclical and act on both daily (diurnal) and seasonal time scales.

As water flows through the wetland, energy gains and losses drive the water temperature towards a balance point temperature, at which energy gains equal energy losses. This results in a longitudinal change in water temperatures as the system trends towards the balance point. The balance point temperature may be warmer or cooler than the influent water temperature, depending on the relative magnitude of the energy flows.

Because temperature exerts a strong influence on some chemical and biological processes, it is important to wetland design. In cold climates, freezing of the wetland may be an operational concern. Successful design requires that forecasts be made for expected or worst-case operating conditions, which implies prediction rules and equations. This chapter reviews the data on treatment wetland water temperatures, and explores the tools available to wetland designers to predict water temperatures that result from energy flows within treatment wetlands.

The water temperature in treatment wetlands is of interest for several reasons:

1. Temperature modifies the rates of several key biological processes.
2. Temperature is sometimes a regulated water quality parameter.
3. Water temperature is a prime determinant of evaporative water loss.
4. Cold-climate wetland systems have to remain functional in subfreezing conditions.

In the first instance, there is extensive literature supporting the strong effect of temperature on microbial nitrogen processing, with doubling of rates over a temperature range of about 10°C. In the second case, cold-water fishes, such as salmonids, are sensitive to water temperature, and cannot survive or breed in warm environments. In the third case, net water loss (and associated increases in total dissolved solids) is a detriment in arid climates, where water rights and water return credits are of increasing importance. Additionally, water temperature is strongly connected to evapotranspiration, which in turn is a major factor in the water budget for the wetland. Finally, freezing of the wetland can create operational problems in cold-climate applications unless the system is designed to avoid freeze-up failure.

4.1 WETLAND ENERGY FLOWS

The energy flows that determine water temperature and the associated evaporative losses are shown in Figure 4.1. These processes are driven and dominated by solar radiation. Incoming solar radiation is partially reflected, with the remainder intercepted by the vegetative canopy and water column. Solar radiation intercepted by the vegetative canopy drives transpiration in plants. The remaining solar radiation is absorbed by the wetland water, and drives evaporation. The combined water loss is termed *evapotranspiration*, and is commonly abbreviated as *ET*.

Convection and diffusion carry water away from the surface, and transfer heat from the air to the wetland. The driving force for convective and diffusive heat transfer is the temperature difference between the wetland and the air above. For water vapor transport, the driving force is the water partial pressure difference between the wetland and the air above. Additionally, heat is radiated from the wetland. Heat may also be transferred from soils to the wetland, but that contribution is usually very small. The net effect of these processes will be a difference between the sensible heats of incoming and outgoing water flows.

Wetland energy flows are the proper framework to interpret and predict not only evaporative processes, but also wetland water temperatures. The energy balance equations involve time-step calculations, and are in general only amenable to computer spreadsheets. However, those calculations are now available from Internet sources, and the wetland designer can readily use this approach. The required input information consists of meteorological information. There are many versions of the energy balance equations that have been put forth, and the interested reader may pursue details in the literature, including the comparative study of *ET* predictive methods for a Florida treatment wetland (Abtew and Obeysekera, 1995). A brief summary of the model will serve to explain these data needs.

ENERGY BALANCE TERMS

Here the methods for calculating each of the quantities in the wetland energy balance are illustrated. The magnitudes of the various energy flows are given in Table 4.1, for FWS wetlands near Phoenix, Arizona (Kadlec, 2006c), in the balance condition. These wetlands were large enough to consider as driven by regional climatic variables. However, freezing conditions are virtually nonexistent at that location. Cold climate wetland considerations are considered in subsequent sections, as are modifications for HSSF systems.

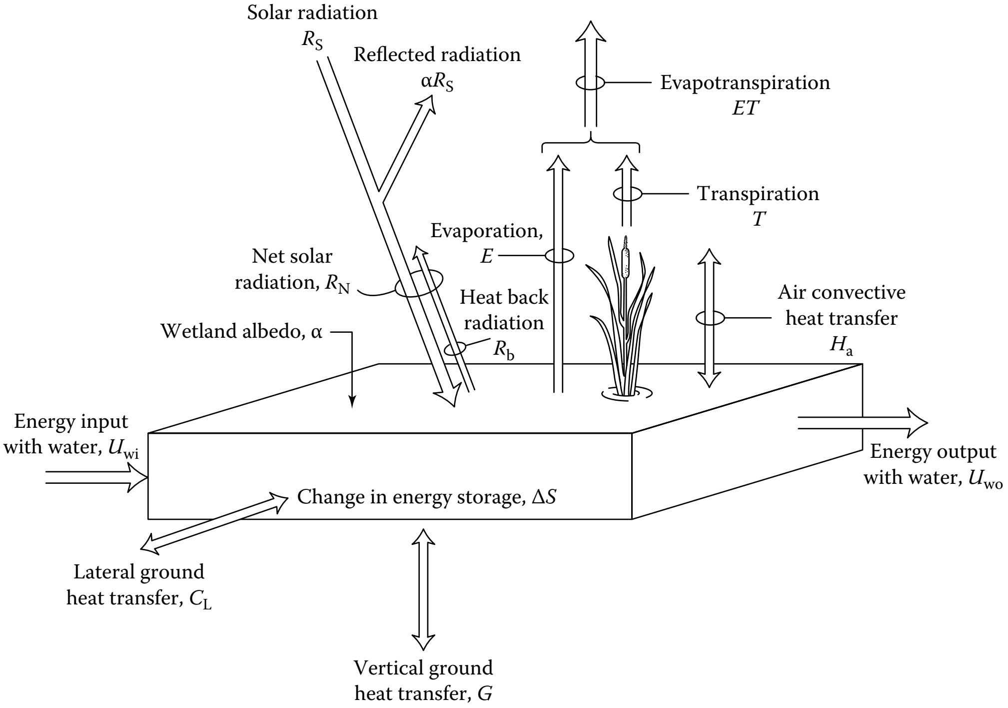


FIGURE 4.1 Components of the wetland energy balance. (From Kadlec and Knight (1996) *Treatment Wetlands.* First edition, CRC Press, Boca Raton, Florida.)

The system for the energy balance is here taken to be the wetland water body and the associated phytomass (Figure 4.1).

Energy Inputs – Energy Outputs = Change in Energy Storage

$$[R_N + H_a + U_{wi}] - [\lambda_m \rho ET + U_{wo} + G + C_L] = \Delta S \qquad (4.1)$$

where

C_L = lateral heat loss to ground, MJ/m²·d
G = vertical conductive loss to ground, MJ/m²·d
ET = water lost to evapotranspiration, m/d
H_a = convective transfer from air, MJ/m²·d
R_N = net radiation absorbed by wetland, MJ/m²·d
ΔS = energy storage change in the wetland, MJ/m²·d

TABLE 4.1
Heat Budget Elements (MJ/m²·d) for a Portion of a FWS Wetland in Phoenix, Arizona, in the Balance Condition

Month	Radiation Net In	Heat Gain from Air	Sensible Heat from Water	Surface Flux from Ground	Total In	Heat Loss from *ET*	Thermal Back Radiation	Total Out
Jan	10.5	0.4	0.0	0.2	**11.2**	4.7	6.4	**11.2**
Feb	13.2	–0.1	0.0	0.1	**13.1**	6.5	6.6	**13.1**
Mar	16.7	–0.2	0.0	0.0	**16.5**	9.7	6.8	**16.5**
Apr	20.4	0.7	0.0	–0.2	**20.9**	13.9	7.0	**20.9**
May	22.9	2.1	0.0	–0.3	**24.8**	17.8	6.9	**24.8**
Jun	23.9	3.3	0.0	–0.3	**26.9**	20.1	6.8	**26.9**
Jul	22.9	3.6	0.0	–0.2	**26.3**	19.8	6.5	**26.3**
Aug	20.2	3.1	0.0	–0.1	**23.1**	16.9	6.2	**23.1**
Sep	16.5	2.2	0.0	0.0	**18.7**	12.5	6.2	**18.7**
Oct	13.0	1.5	0.0	0.2	**14.6**	8.4	6.2	**14.6**
Nov	10.4	1.1	0.0	0.3	**11.8**	5.5	6.3	**11.8**
Dec	9.5	0.9	0.0	0.3	**10.7**	4.3	6.3	**10.7**

Note: The hydraulic loading rate is 15 cm/d.

Source: From Kadlec (2006c) *Ecological Engineering* 26: 328–340. Reprinted with permission.

U_{wi} = energy entering with water, MJ/m²·d
U_{wo} = energy leaving with water, MJ/m²·d
λ_m = latent heat of vaporization of water, MJ/kg (2.453 MJ/kg at 20°C)
ρ = density of water, kg/m³

It is informative to examine these terms, with a view to understanding the magnitude of the various heat fluxes.

Solar Radiation

The net incoming radiation reaching the surface of the wetland may be calculated through a series of steps which estimate the absorptive and reflective losses from incoming extraterrestrial radiation, R_A, shown in Figure 4.1. The amount of radiation which makes it through the outer atmosphere is solar radiation:

$$R_S = \left(0.25 + 0.5\frac{S}{100}\right)R_A \tag{4.2}$$

where
R_A = extraterrestrial radiation, MJ/m²·d
R_S = solar radiation, MJ/m²·d
S = percent daily sunshine

Solar radiation (R_S) is the quantity reported by the several climatological data services as discussed below. The data scatter about an annual sinusoidal trend (Figure 4.2). The upper limit of the data envelope represents cloud-free conditions (S = 100), and individual days may have lesser amounts of incoming radiation.

A fraction α, the wetland albedo, of this radiation is reflected by the wetland. A value of $\alpha = 0.23$ is commonly used for green crops (ASCE, 1990). Priban *et al.* (1992) present seasonally variable values for wetlands, with summer values of 0.18–0.22, and an autumn value of 0.10.

Back Radiation (Radiative Heat Loss)

Net outgoing long wave (heat) radiation is computed based on atmospheric characteristics of cloud cover, absolute temperature, and moisture content:

$$R_b = \left[0.1 + 0.9\left(\frac{S}{100}\right)\right]\left[0.34 - 0.139\sqrt{P_w^{sat}(T_{dp})}\right]\sigma(T+273)^4 \tag{4.3}$$

where
R_b = net outgoing long wave radiation, MJ/m²·d
$P_w^{sat}(T_{dp})$ = water vapor pressure at the dew point, kPa
T = air temperature, °C
σ = Boltzmann's constant = 4.903×10^{-9} MJ/m²·d

In combination, the net incoming radiation is:

$$R_N = 0.77R_S - R_b \tag{4.4}$$

For example, net radiation at Phoenix ranges from (9.5 × 0.77 − 6.3) = 1.0 MJ/m²·d in December, to (23.9 × 0.77 − 6.8) = 11.6 MJ/m²·d in June (see Table 4.1).

Convective Losses and Gains to Air

Although lumped together in Equation 4.1, there are two major and distinct components of heat exchange with air. Wind blows through the wetland plant canopy, and either warms or cools the leaves. In the process, it removes the water transpired through the leaves. Secondarily, this air also may heat or cool the water or gravel bed underlying the canopy.

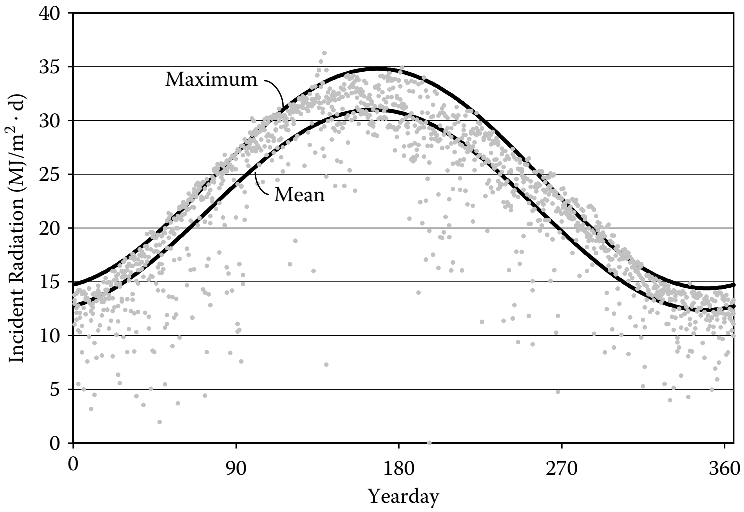


FIGURE 4.2 Solar radiation as a function of season for Phoenix, Arizona. Mean and maximum trendlines are shown, along with data from 1995–1999.

The relative proportions depend upon the extent of vegetative cover, and the relative areas of leaves and water (bed). The effect in the canopy is to control transpiration, whereas the effect in the wetland below is to control evaporation and water temperature.

Accompanying the heat transfer in the canopy, there will be a corresponding mass transfer of water vapor from the leaves to the air passing through. In FWS, there will be a corresponding mass transfer of water vapor from the water surface to the air. However, in HSSF systems, this transfer from water is blocked by dry surface media and also mulch, if used.

Calculations utilize the known relations between the transfer rates and wind speed. For instance, according to ASCE (1990), the vapor flow is calculated as a mass transfer coefficient times the water vapor pressure difference between the water or leaf surface and the ambient air above the wetland:

$$ET = K_e[P_w^{sat}(T_w) - P_{wa}] = K_e \Delta P_w \tag{4.5}$$

where

K_e = water vapor mass transfer coefficient, m/d·kPa
P_{wa} = ambient water vapor pressure, kPa
$P_w^{sat}(T_w)$ = saturation water vapor pressure at T_w, kPa
T_w = water temperature, °C

Typically, the amount of water in the ambient air is a known quantity, calculated as the relative humidity times the saturation pressure of water at the ambient air temperature:

$$P_{wa} = RH \cdot P_w^{sat}(T_{air}) \tag{4.6}$$

where

RH = relative humidity, fraction
T_{air} = air temperature, °C

The water transport coefficient has been found to be a linear function of the wind velocity, the following correlation being one of several in common use (ASCE, 1990):

$$K_e = \frac{(4.82 + 6.38u)}{\lambda} = (10^{-3})(1.965 + 2.60u) \tag{4.7}$$

where

u = wind speed at two meters elevation, m/s
$\lambda = \rho\lambda_m$ = volumetric latent heat of vaporization of water (2,453 MJ/m^3)

The convective heat transfer from the water to the air is likewise represented as a heat transfer coefficient times the temperature difference:

$$H_a = U_{air}\ [T_w - T_{air}] = U_{air}\ \Delta T \tag{4.8}$$

where

U_{air} = heat transfer coefficient, MJ/m^2·d·°C

The relation between heat and mass transfer in the air–water system has resulted in an accurate, calibrated relation between the heat and mass transfer coefficients (ASCE, 1990):

$$U_{air} = \gamma\lambda K_e = (0.0666)(2453)K_e = 163.3K_e \tag{4.9}$$

where

$\gamma = \dfrac{c_p P}{[0.622]\lambda}$ = the psychrometric constant, kPa/°C
γ = 0.0666 at 20° C and 1 kPa and (0.622 = 18/29 = molecular weight ratio of water to air)
c_p = heat capacity of air, MJ/kg°C
P = ambient air pressure, kPa

thus

$$U_{air} = (0.0666)(4.82 + 6.38u) = 0.321 + 0.425u \tag{4.10}$$

For the Phoenix example, exchanges with air range from slight losses of −0.2 MJ/m^2·d in March, to gains of 3.6 MJ/m^2·d in June (Table 4.1). The corresponding heat transfer coefficients were U_{air} = 0.60 ± 0.07 MJ/m^2·°C·d. For the NERCC, Minnesota HSSF wetlands, U_{air} = 0.31 ± 0.03 MJ/m^2·d·°C (Kadlec, 2001b). These values are consistent with the widely accepted value of the heat transfer coefficient in stagnant air above evaporating vegetated surfaces, which is U_{air} = 0.37 MJ/m^2·d·°C (ASCE, 1990). Crites et al. (2006) provide best judgment estimates of U_{air} = 0.13 MJ/m^2·d·°C for dense marshes, 0.86 for open water in still air, and 2.15 for windy conditions in open water.

The energy exchange between water and air in winter in cold climates requires more detailed calculations involving the insulating properties of mulches, ice, and snow. That situation will be discussed separately below.

Conduction Losses and Gains from Soils

In general, lateral energy transfers, horizontally from the wetland edges, are small enough to be negligible. Lateral losses at the Grand Lake, Minnesota, wetland were found to be 0.001–0.003 MJ/m^2·d.

The vertical energy gains and losses from soils below the water are also usually negligible compared to radiation and ET during summer, but are of considerable importance in winter, when they are the only gains. Approximate calculations may be based on the vertical temperature gradient below ground:

$$G = k_g\left(-\frac{dT}{dz}\right) \tag{4.11}$$

where

G = energy gain, MJ/m^2·d
k_g = thermal conductivity of ground, MJ/m·d·°C
T = soil temperature, °C
z = vertical distance upward, m

The thermal conductivities of soils vary with type, with a typical range of 30–190 kJ/m°C·d (Table 4.2). The maximum vertical temperature gradients below treatment wetlands have

TABLE 4.2
Thermal Conductivities of Wetland Solid Materials

Material	Thermal Conductivity (MJ/m·d·°C)
Air	0.0021
Milled peat	0.0043
Granular peat	0.0053
Dry litter (straw)	0.009
New snow	0.007
Dry LECA	0.010
Wet LECA	0.015
Old snow	0.022
Dry gravel	0.026
Dry sand	0.030
Soil	0.045
Water	0.051
Saturated peat	0.052
Clay	0.112
Dry sand	0.152
Ice	0.190

Note: These are generic materials with considerable variability in property values, and the numbers are therefore approximate.

been measured to be in the range of 5–15°C/m, decreasing upward in the winter, and decreasing downward in summer. Accordingly, the heat additions (winter) or losses (summer) reach extremes of 0.15–2.9 MJ/m²·d.

The vertical conduction process has been modeled as transient heat conduction, and fits data quite well for FWS and HSSF systems (Priban *et al.*, 1992; Kadlec, 2001b). The temperature profiles $T(z, t)$ in the (unfrozen) soils below a wetland are governed by the unsteady-state heat conduction equation, together with the boundary condition of a fixed temperature mean annual temperature, a constant at deep locations:

$$\frac{\partial^2 T}{\partial z^2} = \frac{1}{\alpha}\frac{\partial T}{\partial t} \tag{4.12}$$

$$T(\infty, t) = T_s \tag{4.13}$$

For a sinusoidal surface temperature, the solution to this periodic, dynamic heat balance is (Priban *et al.*, 1992):

$$T(z,t) = T_s + A\exp\left(-\frac{z}{H}\right)\cos\left[\omega(t - t_{max}) - \frac{z}{H}\right] \tag{4.14}$$

where

$$H = \sqrt{\frac{2\alpha}{\omega}} \tag{4.15}$$

$$\alpha = \frac{k}{\rho_s c_s} \tag{4.16}$$

and

A = amplitude of surface temperature cycle, °C
c_s = soil heat capacity, MJ/kg·°C
k = soil thermal conductivity, MJ/m·d·°C
t = time, Julian day
t_{max} = time of maximum surface temperature, Julian day
T = temperature, °C
T_s = mean annual temperature of the soil surface, °C
z = vertical depth, m
α = thermal diffusivity of soil, m²/d
ρ_s = soil density, kg/m³
ω = annual cycle frequency = 2 /365 = 0.0172 d⁻¹

The penetration depth (H) is the depth at which the mean annual temperature swing is 63.2% of that at the soil surface (A). The heat flux into the water from the soil is then:

$$G = \left[\frac{kA}{H}\right]\cdot\left[\cos\left(\omega(t - t_{max})\right) - \sin\left(\omega(t - t_{max})\right)\right] \tag{4.17}$$

It may be shown that the heat flux (G) achieves a maximum 46 days (one eighth of an annual cycle) before the day of minimum water temperature, which is also 136 days after the day of maximum water temperature. It may also be shown that the total heat gain from the soil over the 182-day heating half cycle (G_{half}) is:

$$G_{half} = \left(2\sqrt{2}\right)\frac{kA}{\omega H} \tag{4.18}$$

The maximum daily heat gain may be shown to be a factor $\pi/2 = 1.57$ times greater than the average rate over the heating half of the year.

This model provides an accurate description of the temperature gradients below the Grand Lake and NERCC, Minnesota, treatment wetlands (Kadlec, 2001b), as well as the Jackson Meadow, Minnesota, and Houghton Lake, Michigan, treatment wetlands (Table 4.3). In addition to the sinusoidal surface water temperature parameters, only one further constant is needed, the penetration depth (H).

Heating or Cooling of the Water

As water passes through the treatment wetland, it may either cool or warm, depending on meteorological conditions. The energy associated with the water (sensible heat) is a relative quantity, requiring a reference temperature:

$$U = \rho\, c_p\, Q(T_w - T_{ref}) \tag{4.19}$$

where

c_p = heat capacity of water, MJ/kg·°C
Q = water flow, m³/d
T_w = water temperature, °C

TABLE 4.3
Regression Parameters for the Under-Wetland Soil Temperature Heat Conduction Model

Parameter	NERCC 1, Minnesota HSSF	NERCC 2, Minnesota HSSF	Grand Lake, Minnesota HSSF	Jackson Meadow, Minnesota HSSF	Houghton Lake, Michigan FWS
Data years	4	4	4	2	4
Number of depths	4	4	4	4	5
Soil	Mineral	Mineral	Mineral	Mineral	Wet peat
Surface temperature amplitude, °C	8.23	8.23	8.02	10.11	8.15
Surface temperature maximum, Julian day	213	213	217	219	195
Penetration depth, m	2.05	2.24	2.17	0.61	0.95
Thermal diffusivity, m^2/d	0.0361	0.0432	0.0407	0.0032	0.0078
Correlation coefficient, R^2	0.87	0.89	0.88	0.92	0.89
Upward heat flux maximum, Julian day	350	349	353	356	332
Maximum heat flux, MJ/m^2·d	0.274	0.250	0.250	1.189	0.772
Half-year heat gain, MJ/m^2	31.8	29.1	28.9	138	89.8

T_{ref} = reference temperature, °C
U = energy flow with water, MJ/d
ρ = density of water, kg/m^3

The sensible heat increase or decrease from inlet to outlet, per unit area of wetland, is:

$$\Delta U = \rho c_p q(T_{wo} - T_{wi}) \tag{4.20}$$

where
q = hydraulic loading rate, m/d
T_{wi} = inlet water temperature, °C
T_{wo} = outlet water temperature, °C

The energy associated with a 5°C increase in water temperature, at a hydraulic loading rate of 5 cm/d, is 1.04 MJ/m^2·d.

Changes in Storage: Thermal Inertia

Energy is absorbed as the entire wetland heats up, or released as it cools down. Maximum seasonal rates of temperature change are of the order of 0.5°C/d. The energy absorbed in increasing the wetland temperature is:

$$\Delta S = \rho c_p h_w \left(\frac{dT}{dt}\right) \tag{4.21}$$

where
h_w = water depth, m
ΔS = stored energy increase in one day, MJ/m^2·d
dT/dt = water temperature increase rate, °C/d

The heat capacity of the wetland, at a depth of 0.45 m, is (4.182)(0.45) = 1.88 MJ/m^2·°C. The energy associated with a 0.5°C/d increase in mean FWS wetland water temperature is 0.94 MJ/m^2·d.

A HSSF wetland has greater thermal inertia, or storage potential, because of the presence of the gravel matrix. The heat capacity of the wetland is comprised of water and gravel contributions:

$$(\rho ch)_{wetland} = [\varepsilon(\rho c)_{water} + (1-\varepsilon)(\rho c)_{gravel}]h \tag{4.22}$$

where
h = depth of the bed, m
ε = porosity of bed, unitless

For a 45-cm deep bed at porosity 0.4, with gravel heat capacity 0.2 times that of water, which is typical of nearly all stone materials:

$$\begin{aligned}(\rho cV)_{wetland} &= [0.4(1{,}000)(4{,}182) + (1-0.4)(2{,}500)(840)](0.45)\\ &= 1.32\ \text{MJ/m}^2\cdot{}^\circ\text{C}\end{aligned}$$

Here the density of the media has been selected as 2.5 times that of water. The maximum energy storage rate is then is 0.66 MJ/m^2·d.

Shoemaker *et al.* (2005) investigated the role of storage on fluctuations in energy balances for FWS wetlands in Florida. They found that the magnitude of changes in stored heat energy generally decreased as the time scale of the energy balance increased. Daily fluxes of stored heat energy accounted for 20% or more of the magnitude of mean daily net radiation for about 40% of their data, whereas weekly fluxes of stored heat were 20% of mean weekly net radiation for about 20% of the same data. Thus, storage plays a role in dampening short-term energy flow variations.

Heat of Vaporization

Evaporated and transpired water require the input of considerable energy to accomplish the phase change from liquid, in the water column or in the leaves of the canopy, to the vapor form in the air above. As indicated in Equation 4.1, this is computed as the specific heat of vaporization times the evapotranspiration rate, $\lambda_m \rho ET$, where λ_m = 2453 MJ/kg. Wetland *ET* varies seasonally, from minimum values in winter to maxima in summer. Peak midsummer *ET* rates range upward from about 5 mm/d, depending upon wetland size. The peak midsummer energy required therefore ranges upward from 12.3 MJ/m²·d. In Phoenix, heat loss to *ET* ranges from 4.3 to 20.1 MJ/m²·d (see Table 4.1). In temperate climates, in winter, *ET* drops to close to zero. The existence of frozen conditions and snow cover requires additional considerations, given below.

4.2 EVAPOTRANSPIRATION

Water losses to the atmosphere from a wetland occur from the water and soil (evaporation, *E*), and from the emergent portions of the plants (transpiration, *T*). The combination of the two processes is termed evapotranspiration (*ET*). This combined water vapor loss is primarily driven by solar radiation for large wetlands, but may be significantly augmented by heat transfer from air for small wetlands. It is governed by the same wetland energy balance equations that describe wetland water temperatures.

Evapotranspiration is the primary energy loss mechanism for the wetland, and serves to dissipate the majority of the energy. In this context, evapotranspiration can be thought of as the cooling system for the treatment wetland. Without the attendant energy loss through the latent heat of vaporization of water, the "wetland" temperature would increase to a hot, desert-like condition since incoming solar radiation could not be effectively dissipated. Although evapotranspiration is best thought of in terms of the wetland energy balance, sometimes only the water volume lost through *ET* is of concern, and the attendant energy flows associated with *ET* can be ignored. As a result, there are a variety of methods to estimate *ET*. Some estimation methods rely on energy balance calculations, while others rely on surrogate measurements.

Methods of Estimation for *E*, *T*, and *ET*

There are several related measurements of lake and wetland water losses. These measurements are not interchangeable, and indiscriminate use can lead to confusion. Information that can be used to estimate *ET* includes the following:

1. Lake evaporation, which is the loss from large, unvegetated water bodies (*E*).
2. Transpiration, which is the loss of water through above-water (or aboveground) plant parts (*T*).
3. Wetland evapotranspiration, which is the loss from vegetated water bodies (*ET*). Vegetation may be rooted or floating, emergent or submergent.
4. Class A pan evaporation, which is the water loss from a shallow pan of specific design, situated on a specified platform (E_A)
5. Evaporation from closed-bottom lysimeters (pans) of varying design (E_P), containing only water. These may be place in stands of emergent vegetation (E_{PV}) or in areas of open water, with or without submergent or floating plants (E_{PO}).
6. Evapotranspiration from closed-bottom lysimeters (pans) of varying design, which contain soil, plants and water (ET_P). These are placed in stands of comparable vegetation.
7. Regional, large scale, water loss computed from meteorological information, for a reference crop and the assumption of standing water or saturated soil surface (ET_o). Computations may follow one of several energy balance methods, such as Penman–Montieth (Monteith, 1981) or Priestley–Taylor (Priestley and Taylor, 1972).

Energy Balance Methods

For large wetlands, the principal driving force for *ET* is solar radiation. A good share of that radiation is converted to the latent heat of vaporization. About half the net incoming solar radiation is converted to water loss on an annual basis. Reported values include: 0.49, (Bray, 1962); 0.47, (Christiansen and Low, 1970); 0.51, (Kadlec *et al.*, 1987); 0.64, (Roulet and Woo, 1986); 0.54, (Abtew, 1996; 2003). If radiation data from the central Florida area are used to test the concept for the Clermont wetland (Zoltek *et al.*, 1979), the value is 0.49.

Equation 4.1 and its variants are widely used in the literature to predict *ET*. Its use is dependent on equations relating the quantities in Equation 4.1 to meteorological and environmental variables. Incoming radiation depends upon latitude, season, and cloud cover. Incident radiation data are typically readily available from weather stations or summary service organizations, such as the National Climatological Data Center (NCDC) in the United States (http://www.ncdc.noaa.gov), which monitors radiation at 237 stations across the country.

Water losses to the atmosphere from a wetland occur from the water and from emergent vegetation. Convective eddies in the air, associated with wind, swirl water vapor and sensible heat from the water and vegetation upward to the bulk of the overlying air mass. The driving force for water transfer into the air is the humidity difference between the water surface (assumed saturated) and the bulk air. This humidity difference is strongly dependent upon water temperature, via the vapor pressure relationship.

One simple *ET* calculation procedure for large regional wetlands was described in the first edition of this book. It is not repeated here because there are now short cuts available to the treatment wetland designer.

The Reference Crop ET_o Spreadsheet Method

For large wetlands, a common assumption is that *ET* may be represented by the reference crop ET_o computation. The Environmental and Water Resources Institute (EWRI) of the American Society of Civil Engineers (ASCE) established a benchmark reference evapotranspiration equation that standardizes the calculation of reference evapotranspiration

(Allen *et al.*, 2000); (http://www.kimberly.uidaho.edu/water/asceewri/). The intent was to produce consistent calculations for reference evapotranspiration for use in agriculture. A spreadsheet program, PMday.xls, is available (Snyder and Eching, 2000; Snyder, 2001). Inputs include the daily solar radiation (MJ/m^2·d), air temperature (°C), wind speed (m/s), and humidity (e.g., dew point temperature (°C) or relative humidity (%)). The program calculates ET_o using the Penman–Monteith equation (Monteith, 1965) as presented in the United Nations FAO Irrigation and Drainage Paper by Allen *et al.* (1998).

This procedure has been calibrated and verified for a green alfalfa crop, with a fetch of at least 100 m. Other cover types may vary, due to changes in albedo and convective transport and other factors. It is critical to recognize that small wetlands will have significantly greater convective heat transfer and, consequently, ET is amplified in small wetlands.

Reference Crop ET_o from Reporting Services

In the United States, arid states provide extensive documentation of ET_o in support of agricultural irrigation, such as the California Irrigation Management Information System (CIMIS, http://wwwcimis.water.ca.gov/cimis/welcome.jsp), the Arizona Meterological Service (AZMET), and the Washington State University Public Agricultural Weather System (PAWS) (http://paws.prosser.wsu.edu/). A comparable system in the United Kingdom is the Meteorological Office Rainfall and Evaporation Calculating System (MORECS) (Fermor *et al.*, 1999). These services provide the results of energy balance model calculations, usually on a daily time step, for current and recent weather conditions. Figure 4.3 shows an example of the annual pattern of ET_o computed for Phoenix, Arizona. Such annual patterns vary with latitude, as indicated in Figure 4.4.

Direct calibrations and checks have been conducted in wetland environments (Abtew, 1996; German, 2000). As a first approximation, $ET = ET_o$ for large FWS wetlands; however, crop coefficients are required for small systems, as shown in Equation 4.23:

$$ET = K_c \cdot ET_o \tag{4.23}$$

where

K_c = wetland crop coefficient, dimensionless

Lafleur (1990) recommended using the energy balance ET_o estimate as the independent variable in linear regression for specific vegetation types. In agriculture, this approach leads to crop coefficients that influence ET at a specific site. This approach has the advantage of retaining the energy balance used in other ecosystems, but modifying it slightly for site-specific circumstances.

Pan Factor Methods (E_A)

The Class A evaporation pan is a convenient reference, because there are many long-term data stations in the United States. The pan is placed on a platform above ground, and therefore evaporates more water than a lake or large wetland. (ASCE, 1990). Each state operates pans at a few stations, and data are reported in Climatological Data, a publication of the National Oceanic and Atmospheric Administration (NOAA), National Climatic Data Center, and available at (http://www.ncdc.noaa.gov).

Wetland evapotranspiration, ET, over at least the growing season, can be approximated as about 0.70–0.85 times Class A pan evaporation, E_A, from an adjacent open site. The Class A pan integrates the effects of many of the meteorological variables, with the notable exception of advective effects. A multiplier of about 0.8 has been reported in several studies, including: northern Utah, (Christiansen and

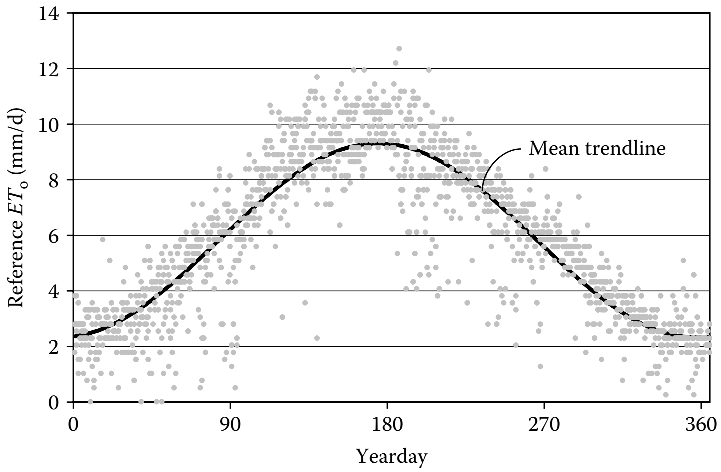


FIGURE 4.3 Reference evapotranspiration (ET_o) as a function of season for Phoenix, Arizona. The mean trendline is shown, along with data from 1995–1999.

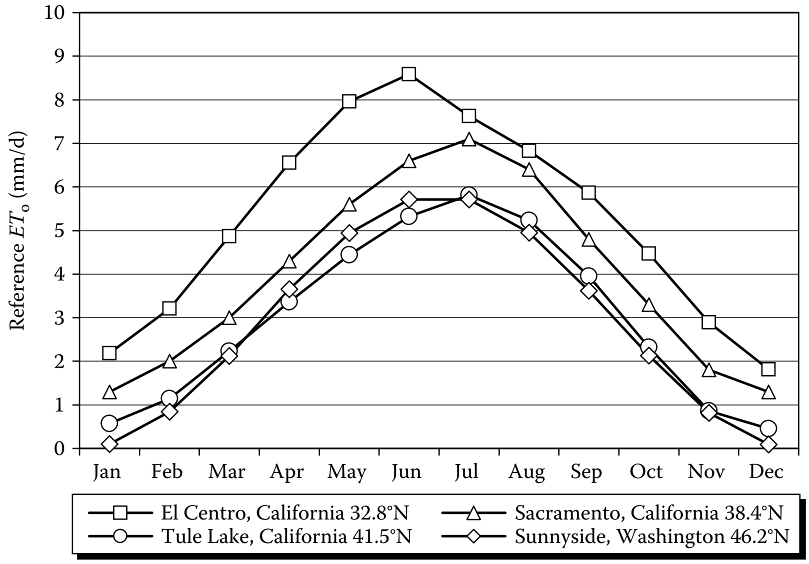


FIGURE 4.4 Reference ET_o as a function of latitude in the western United States.

Low, 1970), western Nevada, (Kadlec *et al.*, 1987), and southern Manitoba (Kadlec, 1986). The stipulation of a time period in excess of the growing season is important, because the short-term effects of the vegetation can invalidate this simple rule of thumb. The effect of climate is apparently small, as the Florida data of Zoltek *et al.* (1979), for a wastewater treatment wetland at Clermont, are represented by 0.78 times the Class A pan data from the nearby station at Lisbon, Florida, on an annual basis. This multiplier is the same as that recommended by Penman (1963) for the potential evapotranspiration from terrestrial systems.

Surface Flow Wetlands

The presence of vegetation retards evaporation in FWS wetlands. This is to be expected for a number of reasons, including shading of the surface, increased humidity near the surface, and reduction of the wind at the surface. The presence of a litter layer can create a mulching effect that reduces open water evaporation (E). The reported magnitude of this reduction is on the order of 50%. A sampling of reduction percentages for open water evaporation includes: (Bernatowicz *et al.*, 1976): 47%; (Koerselman and Beltman, 1988): 41–48%; (Kadlec *et al.*, 1987): 30–86%. However, these data should not be interpreted as meaning that the wetland conserves water, because transpiration (T) can more than offset this reduction.

With plant transpiration offsetting reductions in open-water evaporation, large FWS wetland evapotranspiration and lake evaporation are roughly equal. Roulet and Woo (1986) report this equality for a low arctic site, and Linacre's (1976) review concludes: "In short, rough equality with lakes is probably the most reasonable inference for bog evaporation." Eisenlohr (1966) found that vegetated potholes lost water 12% faster than open water potholes, but Virta (1966) (as cited by Koerselman and Beltman, 1988) found 13% less water loss in peatlands. There is a seasonal effect that can invalidate this general observation in the short term.

The seasonal variation in evapotranspiration shows the effects of both radiation patterns and vegetation patterns. The seasonal pattern of evapotranspiration resembles the seasonal pattern of incoming radiation. During the course of the year, the wetland reflectance changes, the ability to transpire is gained and lost, and a litter layer fluctuates in a mulching function. Agricultural water loss calculations include a crop coefficient to account for the vegetative effect. This is in addition to effects due to radiation, wind, relative humidity, cloud cover, and temperature, and may be viewed as the ratio of wetland evaporation to lake evaporation. The result is a growing season enhancement, followed by winter reductions.

The type of vegetation is not a strong factor in determination of water loss for large, regional wetlands. Bernatowicz *et al.* (1976) found relatively small differences among several reed species, including *Typha*. Koerselman and Beltman (1988) similarly found little difference among two *Carex* species and *Typha*. Linacre (1976) concludes: " ... it appears that differences between plant types are relatively unimportant ... " More recently, Abtew (1996) operated vegetated lysimeters for two years in marshes with three vegetation types: (1) *Typha domingensis*; (2) a mixture including *Pontederia cordata, Sagittaria latifolia,* and *Panicum hemitomon*; and (3) submerged aquatics *Najas guadalupensis* and *Ceratophyllum demersum*. The annual average water losses (ET_P) were 3.6, 3.5, and 3.7 mm/d, respectively.

Subsurface Flow Wetlands

When the water surface is below ground, a key assumption in the energy balance approach is no longer valid: the transfers of water vapor and sensible heat are no longer similar. Water vapor must first diffuse through the dry layer of gravel, and then be transferred by swirls and eddies up through the vegetation to the air above the ecosystem. Heat transfer to the water must now pass through a porous media in addition to the eddy transport in the air for convective transport, or in addition to radiative transport to the gravel surface. The heat storage capacity of the media is also directly involved because it is in the water. The energy balance approach is still valid, but there are no estimates of the transport coefficients within the porous media. It is therefore necessary to rely on wetland-specific information.

Water budgets were used by Bavor *et al.* (1988) to estimate HSSF gravel bed wetland ET for 400 m^2 wetlands in New South Wales, Australia. The correlations with pan measurements were (mm/d):

Gravel (no plants)

$$ET = 0.0757\, E_A - 0.028 \text{ mm/d} \quad R^2 = 0.15 \quad 12°C < T_{air} < 25°C \tag{4.24}$$

Cattails/Gravel (*Typha* spp.)

$$ET = 1.128\, E_A + 0.072 \text{ mm/d} \quad R^2 = 0.72 \quad 12°C < T_{air} < 25°C \tag{4.25}$$

Bulrush/Gravel (*Schoenoplectus* spp.)

$$ET = 0.948\, E_A - 0.027 \text{ mm/d} \quad R^2 = 0.93 \quad 12°C < T_{air} < 25°C \tag{4.26}$$

Comparing the gravel (no plants) ET results (Equation 4.24) to the vegetated (*Typha* and *Schoenoplectus)* systems (Equations 4.25 and 4.26) clearly shows the strong influence of plant transpiration on ET in HSSF wetlands. The gravel effectively cuts off almost all of the evaporative component. Also note that $E_A = 1.25\ ET_o$, so that the annualized crop coefficients (K_c in Equation 4.23) are 1.41 for cattails and 1.19 for bulrushes.

George *et al.* (1998) measured ET in HSSF wetlands at Baxter, Tennessee, 6.0 m^2 in area and vegetated with *Schoenoplectus validus*. Water loss was reported as 1.2 times E_A for healthy vegetation, but drastically less for heavily damaged vegetation. Noting that $E_A = 1.25\ ET_o$, the annual average crop coefficient (K_c) for the Baxter project is estimated to be 1.5.

Fermor *et al.* (1999) investigated ET losses from wastewater reed beds (Himely, United Kingdom, 864 m^2) and runoff reed beds (Teeside International Nature Reserve, United Kingdom), and computed four types of crop coefficients, based upon different methods of determination of ET_o. The regional estimate of ET_o was based upon the assumption of the Penman–Montieth equations, as utilized by the Meteorological Office Rainfall and Evaporation Calculating System (MORECS) in the United Kingdom, calibrated to grass systems on a 40 km × 40 km grid. Results for the Himely HSSF system after maturity are shown in Table 4.4. Water losses are greater than ET_o by a considerable margin, especially in the autumn.

TABLE 4.4
Crop Coefficients for the Himely, United Kingdom, System for 1996

Month	ET (mm/d)	ET_o (mm/d)	K_c
April	1.38	1.81	0.76
May	2.41	2.69	0.90
June	3.84	3.10	1.24
July	4.99	3.10	1.61
August	6.19	2.86	2.16
September	6.30	1.86	3.38
October	2.96	1.49	1.98
Season	**4.01**	**2.42**	**1.66**

Source: Data from Fermor *et al.* (1999) In *Nutrient Cycling and Retention in Natural and Constructed Wetlands.* Vymazal (Ed.), Backhuys Publishers, Leiden, The Netherlands, pp. 165–175.

Size Effects on *ET*

Because many constructed water treatment wetlands tend to be small, it is reasonable to enquire at what size this effect becomes important. There is very little information available on the size effect. The Koerselman and Beltman (1988) study was on a wetland of "less than one hectare," and displayed no large differences from similar studies on larger wetlands. Studies at Listowel, Ontario (Herskowitz, 1986), indicated that lake evaporation was a reasonable estimator of wastewater wetland evapotranspiration for wetlands that aggregated about 2 ha. However, as size is decreased, the advective air energy terms in the energy balance become important at some point, and regional methods are no longer adequate. Ratios to pan and lake evaporation, and to radiation would not be expected to hold.

The use of energy balance information to estimate regional wetland ET is predicated on the assumptions of uniform, equilibrated water temperature, and negligible effects of energy contributions from the air passing through the canopy. There are consequently two factors that may increase water losses from treatment wetlands, in comparison to large scale wetlands in the same locality. The first is the potential for incoming warm water to evaporate to a greater extent than regional waters at ambient conditions. This enhancement is greatest at the point of entry, and diminishes along the flow direction. This effect is more fully discussed next; here, it is noted that the change in water temperature to ambient values (95%) typically occurs in about three or four days' nominal travel time for a FWS wetland. A typical detention time for

FWS systems is seven days. Therefore, for warm incoming waters, enhanced *ET* may be expected over the majority of the flow length.

The second factor has to do with the microclimate created by the wetland. Small wetlands are subject to the "clothesline" and "oasis" effects, in which warm dry air can contribute to heat input and to water loss, well in excess of the loss driven by radiation alone. Indeed, this is the entire basis for the Danish willow systems, which are zero-discharge SSF wetlands (Gregersen and Brix, 2001; Brix and Gregersen, 2002; Brix, 2004; Brix, 2006). This effect has also been reported for other FWS and HSSF wetlands. Estimation of the magnitude and distance scale of this effect may be done by considering the energy balance on the air passing through the canopy of the wetland. If the prevailing wind broadsides the wetland, there is convective transfer of heat to the canopy until the air has lost its heat excess over the regional wetland ambient air. Factors such as the leaf area index (LAI), canopy height, and air temperature and humidity influence the energy balance on the air as it moves through the wetland vegetation. Typical wetland widths for the dissipation of the incoming temperature excess and humidity deficit are on the order of 50 to 100 m (Figure 4.5; Brix, 2006).

The crop coefficient K_c represents the ratio of *ET* for a given wetland to potential ET_o, which represents the regional large system that is always wet. Values of K_c greater than 1.0 mean that the wetland is losing more water than predicted from radiation via the energy balance. For instance, Bavor *et al.* (1988) found *ET* enhanced by a factor of two over pan evaporation in an open-water, unvegetated wetland 4 m wide by 100 m long. Typically, additional *ET* losses are the greatest for the smallest systems, namely microcosms and mesocosms. Rozkošný *et al.* (2006) studied water losses from *Phragmites* and *Typha* in 0.2 m^2 SSF mesocosms (essentially potted plants), which contained 3,000–6,000 g dw/m^2 of vegetation. An unvegetated mesocosm with a free water surface (FWS) was the reference. The values of K_c were found to be 5.4 for *Typha*, and 7.3 for *Phragmites*. Mesocosm studies (Snyder and Boyd, 1987) displayed a strong effect of vegetation and its rate of growth (Table 4.5) This is not unexpected, because the plants exhibit strong edge effects in mesocosms, due in large part to canopy overhang for emergent vegetation. However, convective processes are also magnified in mesocosms, and hence even floating plant systems show species differences in water loss rates. For instance, mesocosm studies by DeBusk *et al.* (1983) showed that open water and *Lemna minor* systems had similar annual average water loss (4.5 and 4.1 mm/d, respectively), but *Eichhornia crassipes* was greater (7.5 mm/d). For such small systems, vegetative overgrowth augments meteorological enhancement.

Wetlands with tall vegetation with large leaf area (LAI) will intercept more dry wind, and exhibit larger K_c. Therefore, willows with a height of 3–4 m will exhibit K_c up to 2.5 (Danish systems). And, for HSSF wetlands, no vegetation causes a virtual elimination of *ET* (Equation 4.24). It is clear that most HSSF wetlands are small enough to exhibit enhanced evapotranspiration, compared to regional energy balance estimates.

Timing of ET Losses

The loss of water from the wetland does not occur uniformly over the course of the day, but rather occurs during the daytime hours. This is occasioned by (1) the radiation driving force is only operative during daylight hours, and (2) wind and dry conditions usually also operate during the daytime. As a consequence, *ET* is nearly zero except for a period of about 12 hours at temperate latitudes in summer. During that period, it displays a parabolic curve, with a maximum at

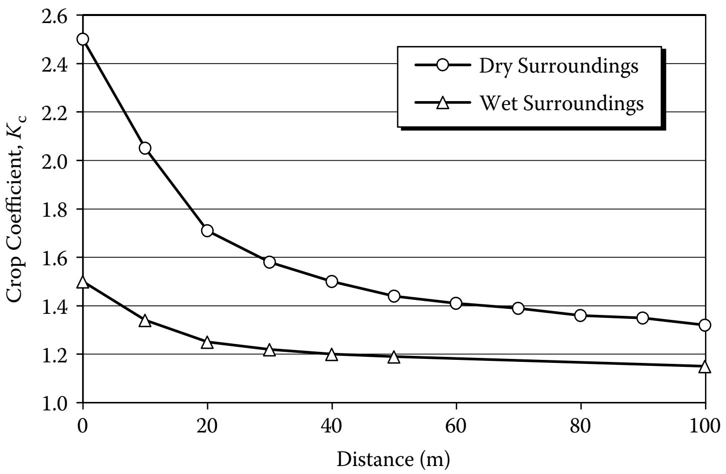


FIGURE 4.5 Enhanced evapotranspiration for small wetlands due to cross-flow winds. K_c is the crop coefficient, or multiplier on regional evapotranspiration for large wetlands. Conditions of wind and humidity are those typical of Denmark in the warm season. (Data from Brix (2006) Course Notes: Onsite treatment of wastewater in willow systems. Aarhus, Denmark, Department of Biological Sciences, Aarhus University.)

TABLE 4.5
Water Loss from Cattail Wetlands

	Open Water E (mm/d)	Low Fertilization		High Fertilization	
		ET (mm/d)	$K_c = ET/E$	ET (mm/d)	$K_c = ET/E$
May	5.6	7.2	1.3	7.6	1.3
June	6.2	9.9	1.6	12.0	1.9
July	4.8	8.5	1.8	12.0	2.5
August	4.8	7.2	1.5	10.4	2.2
September	4.7	5.7	1.2	8.0	1.7
October	3.7	3.8	1.1	5.3	1.4
Season	**5.0**	**7.1**	**1.4**	**9.2**	**1.8**

Note: High fertilization produced peak aboveground biomass of 1,000 g dw/m^2 and LAI = 6.5; low fertilization 500 g dw/m^2 and LAI = 3.5. Means of triplicate 6 m^2 mesocosms.

Source: Adapted from Snyder and Boyd (1987) *Aquatic Botany* 27: 217–227.

about midday, reaching about triple the mean daily ET loss (Scheffe, 1978; Kadlec *et al.*, 1987; Snyder and Boyd, 1987). The result can be strong diel trends in the outflow from the wetland (see Figure 2.5).

TRANSPIRATION: FLOWS INTO THE ROOT ZONE

Vertical flows of water in the upper soil horizon are driven by gravity and by plant uptake to support transpiration. In an aquatic system, without emergent transpiring plant parts, vertical downflow will be driven solely by gravity. Water infiltration flow is then computed from the water pressure (hydraulic head) gradient between the saturated soil surface and the receiving aquifer, multiplied by the hydraulic conductivity of the soil. If the hydraulic conductivity of the soil layers beneath the root zone is very low, then percolation to groundwater is effectively blocked.

In aquatic and wetland systems with fully saturated soils or free surface water, the meteorological energy budget requires the vaporization of an amount of water sufficient to balance solar radiation and convective losses. Some of this vaporization is from the water surface (evaporation); some is from the emergent plants (transpiration). Emergent plants "pump" water from the root zone to the leaves, from which water evaporates through stomata, which constitutes the transpiration loss (Figure 4.6). Water for transpiration must move through the soil to the roots. That movement is vertically downward from overlying waters in most FWS wetland situations, whereas it is directly from the flowing water in HSSF wetlands. In temperate climates, ET ranges from 60 to 200 cm/yr, but is concentrated in that part of the year with greatest solar radiation. Thus, transpiration has the potential to move on the order of one meter per year of water vertically downward to the root zone. This vertical flux of water carries with it the pollutant content of the overlying water, together with soluble materials formed in the root zone. This transpiration-driven pollutant transfer is far greater than the diffusion fluxes (Kadlec, 1999a).

The supply of terrestrial plant nutrients is well known to correlate strongly with this vertical movement of water (Vrugt *et al.*, 2001; van den Berg *et al.*, 2002; Novak and Vidovic, 2003). Novak and Vidovic (2003) state that "It is important that the transpiration flow that drives nutrient transport can be estimated relatively easily ... " Therefore, to understand wetland nutrient removal, it is necessary to separate the processes of wetland evaporation and transpiration.

This situation is well described in the literature (Nobel, 1999), by considering the canopy and water as separate components of the wetland ecosystem for energy budget purposes. Measurements of the two components of ET have shown that shading reduces surface water evaporation, while transpiration continues from the canopy (Kadlec *et al.*, 1987). Herbst and Kappen (1999) report that transpiration accounted for 64 ± 6% of ET in a *Phragmites* stand, measured over a four-year period. Kadlec (2006c) found approximately 70% of ET was due to transpiration in an arid region FWS wetland on an annual basis, but monthly proportions ranged from 45% to 85%.

In a densely vegetated FWS wetland, and in HSSF wetlands, transpiration dominates the combined process of evapotranspiration (Kadlec *et al.*, 1987). The fraction T/ET varies with vegetation density, which in this context is usually characterized by the leaf area index (LAI), defined as the leaf area per unit land/water surface area. Values of the LAI range from less than 1.0 m^2/m^2 in sparsely vegetated systems, to over 5.0 m^2/m^2 in densely vegetated systems (Koch and Rawlick, 1993; Nolte and Associates, 1997; Herbst and Kappen, 1999). The corresponding fractions are $0.5 < T/ET < 0.9$ (Shuttleworth and Wallace, 1985). Figure 4.7 shows the LAI dependence of the T/ET ratio for subtropical conditions (Shuttleworth and Wallace, 1985).

The effects of transpiration and evaporation on wetland pollutant processing in FWS are quite different.

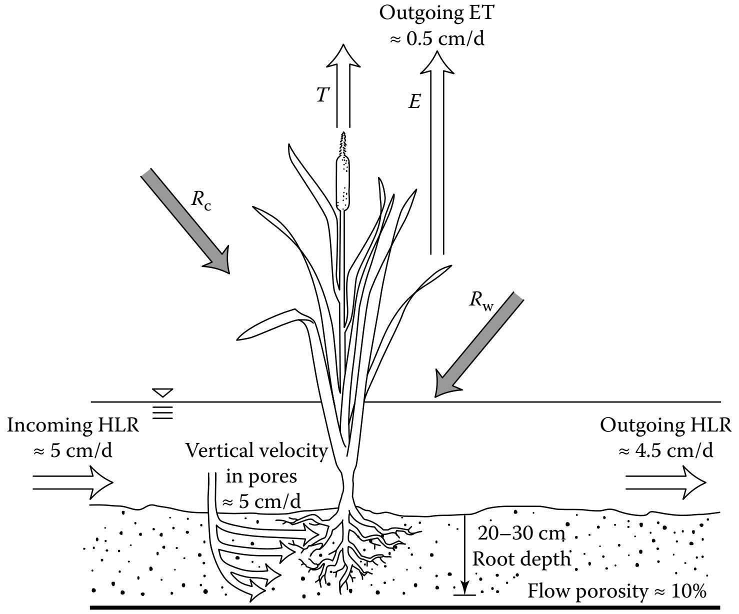


FIGURE 4.6 Transpiration flows create a vertical flux of water that transports phosphorus from the litter-benthic mat zone down into the root zone. The vertical location of water extraction is dependent on the vertical position and density of the imbibing roots.

Transpiration pulls water into the root zone, and into roots, and therefore overcomes transfer resistances. The water loss occurs at the leaves, and hence heat effects are located in the canopy. On the other hand, evaporation concentrates pollutants in the flowing water, and draws the energy directly from the water column, contributing to wetland water cooling. The transpiration flow may be a minor fraction of wetland throughflow in the case of heavily loaded wetlands. For instance, if the hydraulic loading rate is 5 cm/d, and $T = 0.75ET = 0.75 \times 0.5 = 0.375$ cm/d, then $T/q = 7.5\%$. However, for lightly loaded wetlands, transpiration may be much a more important fraction. For instance, if the hydraulic loading rate is 0.5 cm/d, and $T = 0.75ET = 0.75 \times 0.5 = 0.375$ cm/d, then $T/q = 75\%$.

4.3 WETLAND WATER TEMPERATURES

The energy flows that determine water temperature and the associated evaporative losses are shown in Figure 4.1 for a FWS wetland. A treatment wetland may contain one or two thermal regions, depending on water loading (detention time). For long detention times, there is an inlet region in which water temperatures adjust to the prevailing meteorological conditions, and an outlet region in which that adjustment is complete (Figure 4.8). After adjustment, temperature does not change further with distance, or detention time. The value reached is determined by the balance of energy flows and is termed the *balance temperature*. For short detention

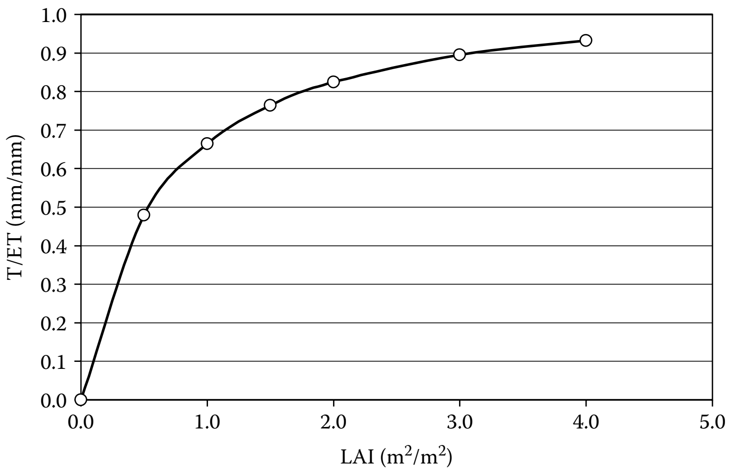


FIGURE 4.7 Fraction transpiration versus leaf area index (LAI) according to the energy partition model of Shuttleworth and Wallace (1985).

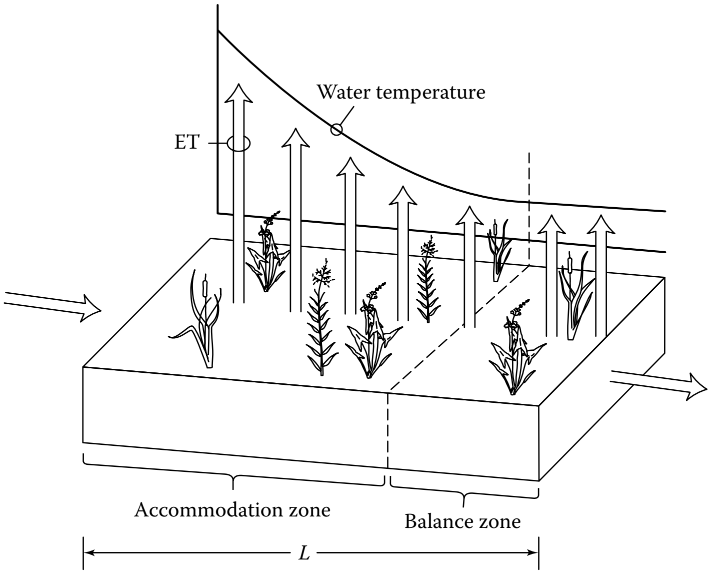


FIGURE 4.8 Gradients in temperature and evapotranspiration in a wetland. (From Kadlec (2006c) *Ecological Engineering* 26: 328–340. Reprinted with permission.)

times, near the wetland inlet, the adjustment may not be completed, and the balance temperature is not reached. In this adjustment or accommodation region, there will be a difference between the sensible heats of incoming and outgoing water flows; in contrast, they are equal in the balance region. In the balance region, sensible heat of the flowing water is therefore not a factor in the energy budget.

To a very rough approximation, wetland water balance temperatures are linearly related to air temperatures during the unfrozen season (Figure 4.9). In winter, the balance point is just above freezing, as long as liquid water is present. However, this approximation is insufficient to support either the design of wetlands for temperature modulation, or for the determination of the temperature effects on microbial processes. Additionally, the incoming water may have quite a different thermal condition, depending upon the type of pretreatment. Lagoon pretreatment leads to water nearly at wetland temperature, whereas activated sludge effluents are likely to be much warmer in winter. Therefore, in many instances, the inlet section of a treatment wetland will contain water that is at a different temperature than the balance point temperature.

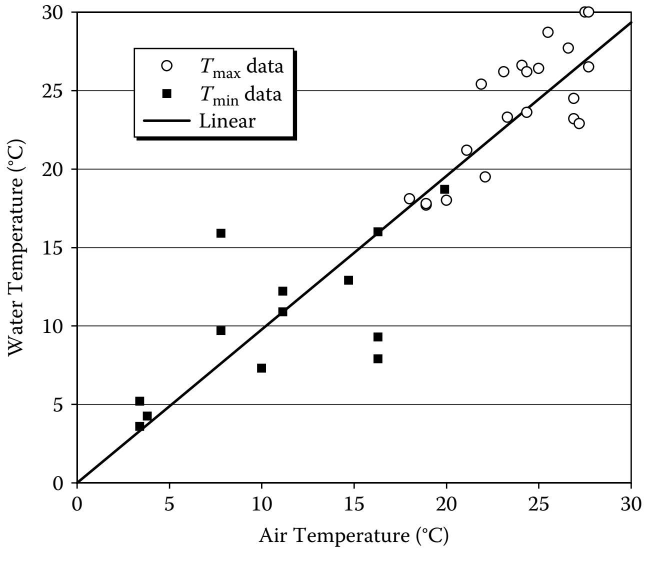


FIGURE 4.9 Relation between annual maximum and minimum water and air temperatures for FWS wetlands. In general, arid climate systems lie below the line, and humid climate systems lie above. $T_w = 0.98T_a$; N = 36; $R^2 = 0.84$; standard error in $T_w = 3.3$°C.

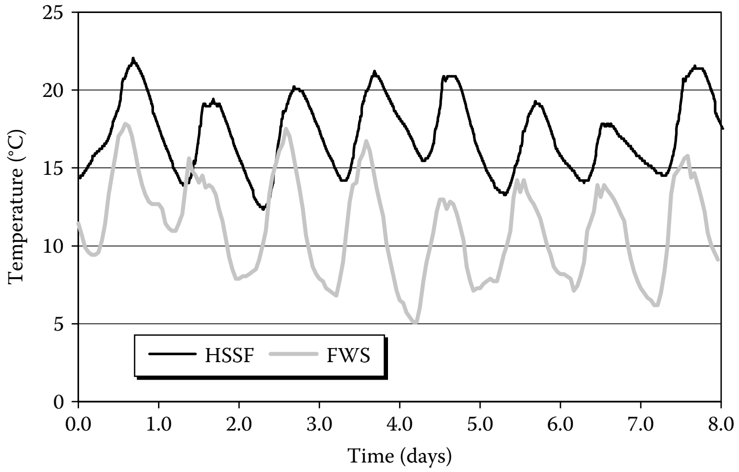


FIGURE 4.10 Diurnal temperature fluctuations in treatment wetlands. The subsurface flow system was treating dairy wastewater (November 21–27). (Tanner, unpublished data). The FWS wetland was treating municipal lagoon effluent (June 1–8). (Kadlec, unpublished data.)

Clearly, simple rules of thumb are not adequate to characterize wetland temperatures. More detail is developed via the observations and models presented below.

Short-Term Cycles

The amplitude of the daily water temperature swing depends on the type of wetland in question, and the type and density of vegetation (Figure 4.10). The general pattern is a marked diurnal swing in water temperature, which can be as large as 8 to 10°C in the warm months. Ordinarily, these daily cycles may be averaged to interpret wetland performance, but there are some exceptions. For instance, daily monitoring at the Tres Rios demonstration project was routinely conducted in the early daylight hours, because of the extreme heat later in the day in southern Arizona (Wass, 1997). Interpretation of the diurnal variation indicated that those morning values were about 2°C lower than the daily average. Determination of the temperature coefficients for microbial processes was therefore based upon adjusted temperatures.

Annual Cycles

The annual cycle of wetland water temperatures in mild to warm climates follows a sinusoidal pattern, with a summer maximum and a winter minimum. In northern climates, the onset of frozen conditions typically is accompanied by under-ice water temperatures of 1–2°C. The sinusoidal model, truncated for frozen conditions, is:

For the unfrozen season ($t_1 < t < t_2$):

$$T_w = T_{avg}\left(1 + A \cdot \cos\left[\omega(t - t_{max})\right]\right) \quad (4.27)$$

For the frozen season ($t_2 < t < t_1$):

$$T_w = T_o \quad (4.28)$$

where

A = fractional amplitude of the sinusoid, unitless
ω = yearly cycle frequency = $2\pi/365 = 0.0172\ d^{-1}$
t = time, Julian day
t_1 = ice-out time, Julian day
t_2 = freeze-up time, Julian day
t_{max} = time of annual maximum temperature, Julian day
T_w = water temperature, °C
T_{avg} = annual average water temperature, °C
T_o = under-ice water temperature, °C

The various quantities associated with this time series model are illustrated in Figure 4.11. Model fits for two example datasets are shown in Figures 4.12 and 4.13. The Imperial, California, FWS cycle does not require truncation, and the weekly data fit has $R^2 = 0.97$. The Grand Lake, Minnesota, HSSF cycle requires truncation, and the daily data fit has $R^2 = 0.94$.

Three parameters are required for Equation 4.27: T_{avg}, A, and t_{max}. Three are also required for Equation 4.28: t_1, t_2, and T_o. Data from several free water surface (FWS) wetlands were regressed to a truncated, sinusoidal time series model (Table 4.6). Data from two to eight years at each site were folded into a composite annual pattern. From this information, it is seen that the time of maximum wetland water temperature is essentially fixed at $t_{max} = 200 \pm 4$ days (mean ± std. dev., N = 14). Data from HSSF systems is likewise well fit by Equations 4.27 and 4.28 (Table 4.7). For these HSSF wetlands, the time of maximum wetland water temperature is at $t_{max} = 210 \pm 6$ days (mean ± std. dev., N = 12). The difference may be attributed to the thermal lag associated with the gravel media in the SSF wetlands. The under-ice temperature is also in a very narrow range of $1 < T_o < 2$°C. It is therefore acceptable to presume an average value of about 1.5°C as an estimation.

The remaining four parameters are site-specific. The treatment wetland designer will be able to find or estimate the

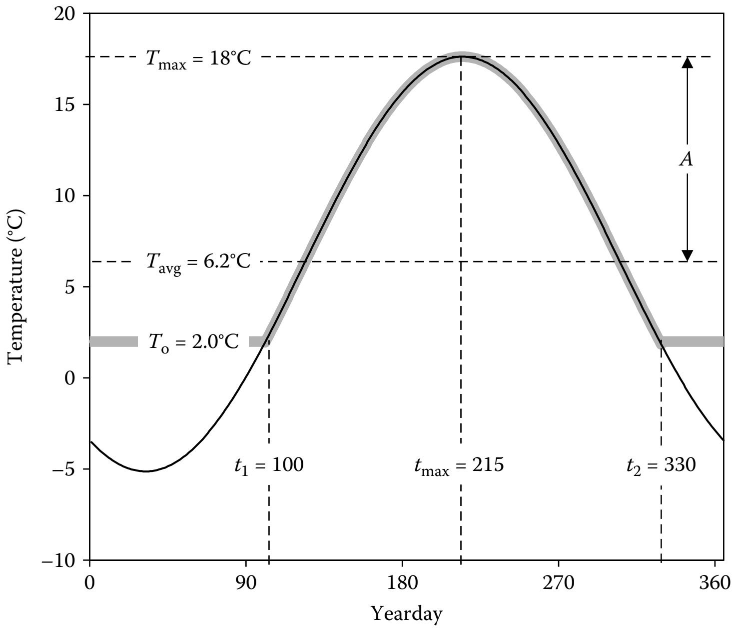


FIGURE 4.11 Sinusoidal model for cyclic annual time series of wetland water temperatures. (*Note:* This example is for purposes of defining terms, and does not represent any real system.)

times of freeze-up and thaw for the site in question. However, there is not a lot of variability for the time of freeze-up for north temperate climates, $t_2 = 332 \pm 21$ days (Table 4.6 mean ± std. dev., N = 10). There is more variation in the time of spring thaw, with $28 < t_2 < 112$ days. Values of A and T_{avg} are given in Tables 4.6 and 4.7 for a number of treatment wetland sites. In qualitative terms, T_{avg} increases and A decreases as the site moves to warmer latitudes. Because of the symmetry of the sinusoid around t_{max}, there is a necessary relation between t_1 and t_2:

$$(t_{max} - t_1) = (t_2 - t_{max}) \tag{4.29}$$

The remaining two parameters, A and T_{avg}, depend upon site climatic conditions. These pertain to the sinusoidal portion of the temperature time sequence, and not to the entire annual profile in the case of truncated profiles. In the case of the truncated annual time series, one further parameter is most conveniently the maximum wetland temperature. The maximum sinusoidal value is then:

$$T_{max} = T_{avg}(1 + A) \tag{4.30}$$

where

T_{max} = maximum wetland water temperature, °C

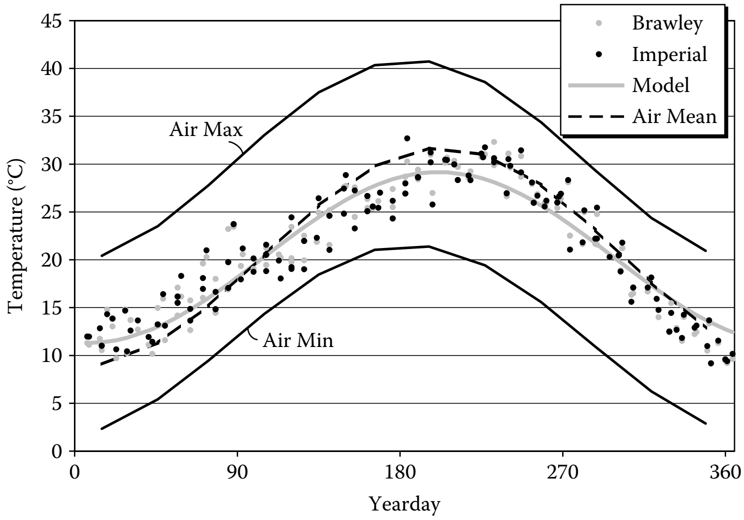


FIGURE 4.12 Annual pattern of water temperatures in the Imperial and Brawley, California, FWS treatment wetlands, compared to air temperatures. Data spans the four-year period of record. (Unpublished data from Imperial, California, Irrigation District.)

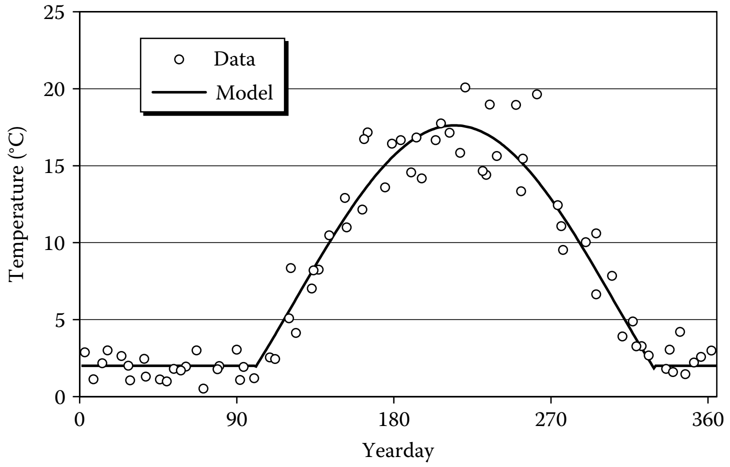


FIGURE 4.13 Annual pattern of water temperatures in the Grand Lake, Minnesota, HSSF treatment wetland.

TABLE 4.6
FWS Water and Air Temperature Regression Parameters

Site	Location	Wetland	T_{mean}	A	$A \cdot T_{mean}$	R^2	T_{max}	T_{min}	T_o	t_{max}	t_1	t_2
Musselwhite	Ontario	Wetland	1.9	8.43	16.0	0.72	18.1	2.0	2.0	206	112	300
Musselwhite	Ontario	Air	−1.0	−19.96	20.0	0.99	18	−20.1	—	205	—	—
Minot	North Dakota	Wetland	5.5	2.55	14.0	0.91	19.5	1.0	1.0	205	90	317
Minot	North Dakota	Air	5.2	3.23	16.9	0.99	22.1	−11.7	—	201	—	—
Listowel	Ontario	3	6.3	1.80	11.3	0.95	17.7	1.5	1.5	204	88	320
Listowel	Ontario	4	6.6	1.69	11.2	0.94	17.8	1.5	1.5	205	86	324
Listowel	Ontario	Air	7.2	2.65	19.1	0.99	18.9	−8.5	—	206	—	—
Brighton	Ontario	Wetland	8.7	1.47	12.7	0.97	20.4	1.0	1.0	206	80	332
Brighton	Ontario	Air	7.2	1.95	13.9	0.99	21.1	−6.8	—	205	—	—
Connell	Washington	Wetland	9.7	0.86	8.3	0.55	18.0	1.5	1.5	201	37	365
Connell	Washington	Air	8.5	1.36	11.6	0.98	20.0	−3.0	—	199	—	—
Hillsdale	Michigan	Wetland	13.5	0.88	11.9	0.98	25.4	2.0	2	196	28	365
Hillsdale	Michigan	Air	8.7	1.51	13.1	0.95	21.9	−4.5	—	209	—	—
Commerce	Michigan	Wetland	11.8	0.89	10.5	0.96	22.3	2.0	2	204	44	365
Commerce	Michigan	Air	10.6	1.19	12.6	0.99	23.3	−2.1	—	202	—	—
Columbia	Missouri	Wetland	14.3	0.68	9.7	0.99	24.1	4.6	—	201	—	—
Columbia	Missouri	Air	13.0	1.04	13.6	0.99	26.6	−0.5	—	201	—	—
Benton	Kentucky	1	13.4	0.73	9.8	0.87	23.2	3.6	—	196	—	—
Benton	Kentucky	2	14.8	0.65	9.6	0.86	24.5	5.2	—	195	—	—
Benton	Kentucky	Air	15.1	0.78	11.8	1.00	26.9	3.4	—	200	—	—
New Hanover	North Carolina	Wetland	18.7	0.48	9.0	0.96	27.7	9.7	—	199	—	—
New Hanover	North Carolina	Air	17.2	0.54	9.3	0.97	26.6	7.8	—	217	—	—
Imperial	California	Wetland	20.2	0.44	11.3	0.97	29.2	11.3	—	201	—	—
Imperial	California	Air	20.3	0.56	11.4	0.95	31.7	8.9	—	204	—	—
Tres Rios	Arizona	Wetland	16.5	0.55	9.1	0.95	25.7	7.3	—	194	—	—
Tres Rios	Arizona	Air	21.3	0.53	11.3	0.87	32.6	10.0	—	202	—	—
ENR	Florida	Wetland	24.4	0.23	5.6	0.77	30.2	18.7	—	196	—	—
ENR	Florida	Air	24.3	0.18	4.4	0.98	28.7	19.9	—	207	—	—

Note: Arranged in order of increasing mean air temperature. ***Important***: T_{mean} and A refer to the sinusoidal portion of the annual time series, and are not the overall annual means for truncated times series sinusoids.

TABLE 4.7
Annual HSSF Wetland Water Temperature Cycle Parameters for Systems in Several Geographic Regions

Site	Latitude	Years	T_{mean} (°C)	Amplitude	Freeze-Up (Julian day)	Thaw (Julian day)	t_{max} (Julian day)	R^2
Haugstein, Norway	60N	5	6.4	3.07	320	100	209	0.94
Grand Lake, Minnesota	47N	4	8.0	2.73	330	100	215	0.94
NERCC 2, Minnesota	47N	4	7.9	2.72	330	100	215	0.96
NERCC 1, Minnesota	47N	4	8.0	2.77	325	100	214	0.95
Minoa, New York	43N	2	10.7	0.91	350	80	217	0.98
Valleyfield 2, Scotland	56N	2	10.0	0.49	N	N	208	0.85
Valleyfield 3, Scotland	56N	2	10.5	0.47	N	N	211	0.85
Valleyfield 4, Scotland	56N	2	10.5	0.45	N	N	211	0.84
Valleyfield 1, Scotland	56N	2	10.6	0.47	N	N	205	0.83
Benton, Kentucky	37N	1	13.9	0.68	N	N	195	0.88
Richmond, NSW, *Schoenoplectus*	34S	2	18.2	0.34	N	N	214	0.86
Richmond, NSW, *Typha*	34S	2	18.3	0.32	N	N	208	0.88
Richmond, NSW, gravel only	34S	2	18.5	0.38	N	N	212	0.86

Note: Systems with freezing conditions all regressed to winter water $T = 2.0$°C, which pertained to the period from freeze-up to thaw. During unfrozen periods, regression was to a sinusoidal pattern. Julian days at southern latitudes are advanced to correspond to northern latitudes.

The sinusoid is then fully defined by choosing A so that $T = T_o$ at $t = t_2$:

$$T_o = T_{avg}\left(1 + A\cos\left[\omega(t_2 - t_{max})\right]\right) \tag{4.31}$$

Solving for A and T_{avg} gives:

$$A = \frac{T_{max} - T_o}{T_o - T_{max}\cos\left[\omega(t_2 - t_{max})\right]} \tag{4.32}$$

and

$$T_{avg} = \frac{T_{max}}{(1 + A)} \tag{4.33}$$

This cyclic model allows quantification of existing data sets, so that information from a variety of wetlands may be compared. It is, however, not predictive, because T_{max} depends upon site climatic conditions.

Predicting Wetland Water Balance Temperatures

The energy balance also determines the equilibrium water surface temperature (Monteith, 1981), but that aspect of the energy balance is not routinely described or reported in connection with ET_o calculations. However, this temperature is easily retrieved, from any energy balance estimate of ET_o. The ET_o loss depends on the difference in water partial pressures between the water or leaf surface and the ambient air above:

$$ET_o = K_e\left[P^{sat}(T_w) - RH \cdot P^{sat}(T_a)\right] \tag{4.34}$$

where

K_e = water vapor mass transfer coefficient, m/d·kPa
$P^{sat}(T_a)$ = saturation water vapor pressure at T_a, kPa
$P^{sat}(T_w)$ = saturation water vapor pressure at T_w, kPa
T_a = air temperature, °C
RH = relative humidity, fraction
T_w = water temperature, °C

Equation 4.34 shows that the water vapor driven off by solar radiation must be convected into the air according to a water partial pressure difference from the water or leaf surface to the ambient air. The water content of the air is determined by both the air temperature and the relative humidity. At high humidity, water temperatures must be high to sustain the mass transfer gradient; conversely, at low humidity, water temperatures are lower.

The air transport coefficient depends on wind speed, and may be represented as a linear function of the wind velocity. For instance, (ASCE, 1990) suggests:

$$K_e = 1.96 + 2.60u \tag{4.35}$$

where
u = wind speed at two meters elevation, m/s

Equations 4.34 and 4.35 combine to give:

$$P^{sat}(T_w) = P^{sat}(T_a) + \frac{ET_o}{(1.96 + 2.60u)} \tag{4.36}$$

The saturation temperature corresponding to a given vapor pressure may be determined from:

$$P^{sat} = 19.0971 - \frac{5349.93}{(T + 273.16)} \tag{4.37}$$

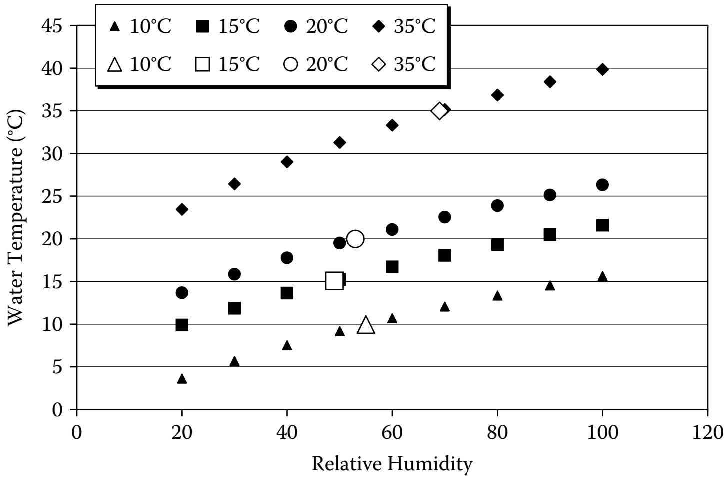


FIGURE 4.14 Variation of wetland balance temperature with relative humidity and air temperature. These four examples use (T_a, ET_o) = (20°C, 5 mm/d); (15°C, 4 mm/d); (10°C, 2.5 mm/d); and (35°C, 8 mm/d), with a wind speed of 1.0 m/s. Open points show the humidity at which the balance temperature equals the air temperature.

Equations 4.36 and 4.37 combine to permit estimation of the balance water temperature. Example calculations show that balance water temperatures are approximately equal to air temperatures for relative humidities of about 50% (Figure 4.14). But, in arid regions water may experience significant evaporative cooling upon transit through the wetland (Kadlec, 2006c).

In some instances, such as densely vegetated wetlands in hot climates, the separate energy balances for the above-water canopy and the water may be needed to obtain a reasonable model for wetland water temperatures (Kadlec, 2006c).

Water Temperature Variability

The deterministic trend expressed in Equations 4.27 and 4.28 represents the central tendency of water temperatures, but there are also stochastic variations. Daily meteorological variations in air temperature, cloudiness, windiness, and relative humidity cause responses in water temperatures, as do changes in incoming water temperatures (see Figures 4.12 and 4.13). Together, these factors create the need to add variability to the trend:

$$T = T_{avg}\left(1 + A \cdot \cos\left[\omega(t - t_{max})\right]\right) + E \tag{4.38}$$

where

E = stochastic contribution to water temperature, °C

The values of E follow a distribution that is nearly normal for either FWS or HSSF wetlands (Figures 4.15 and 4.16). The breadth of the scatter does not change materially during the course of the year, so that E does not depend upon time (t). However, the breadth of the E distribution does depend upon sampling frequency. The standard deviation of the daily Columbia, Missouri, FWS distribution (Figure 4.15) is 2.8°C, whereas for monthly means it is 1.6°C. The standard deviation of the monthly Grand Lake, Minnesota, HSSF distribution (Figures 4.13 and 4.16) is 1.5°C.

Vertical Temperature Stratification

Water density is a function of temperature; with the unusual property that the maximum density is achieved at 4°C (Lide, 1992). Changes in water temperature may result in layers of water with different densities, and partition the water column into discrete density/temperature layers. Thermal stratification is frequently observed in temperate-climate lake systems. Waste stabilization ponds and lagoons, which have depths in excess of 2 m, often exhibit marked stratification during most portions of the year (Torres *et al.*, 2000; Abis, 2002). These phenomena are thoroughly described in the literature on limnology (Wetzel, 2001).

In the summer, solar radiation raises the temperature of the surface water, reducing its density. The less-dense surface water is buoyant relative to the cooler (and denser) water layer underneath. While thermally-induced vertical stratification in lakes is typically thought of in terms of long-term seasonal effects, daily stratification can also occur due to the diurnal fluctuation in solar radiation.

There are three potential regimes for vertical temperature profiles that have been observed in wetlands and shallow ponds. There may be no vertical profile at all, a condition of no thermal stratification. The second situation is no vertical profile during the night, but the development of surface heating during the daytime hours. This is termed diurnal mixing. The third case is the existence of a vertical temperature gradient throughout the entire 24-hour period, called *stratification*.

Breen and Lawrence (1998) suggest that wind speed is the primary determinant for stratification of shallow ponds in subtropical conditions. They suggest that winds less 0.6 m/s lead to stratification, 0.6–2 m/s lead to diurnal mixing, and greater than 2 m/s provide for full mixing.

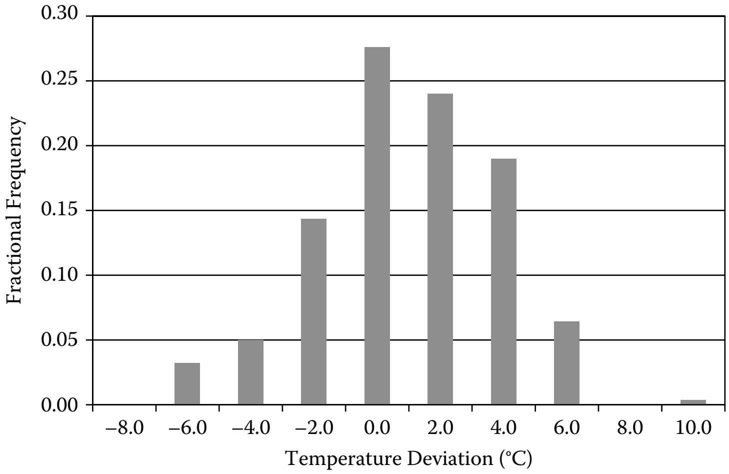


FIGURE 4.15 Deviations of daily temperatures from the sinusoidal trend for the Columbia, Missouri, FWS treatment wetland.

Condie and Webster (2001) present a criterion for stratification based on pond/wetland models and data from a shallow unvegetated Australian billabong. This criterion is based upon the dimensionless group:

$$S = \frac{\rho c_p u^3}{g \alpha h R_N} \tag{4.39}$$

where

c_p = heat capacity of water, 4.182×10^6 J/kg·°C
g = acceleration of gravity, 9.8 m/s^2
h = water depth, m
R_N = net solar radiation, J/m^2·s
S = stratification group, unitless
u = wind speed at 2 m elevation, m/s
α = thermal expansion coefficient of water, 2×10^{-4} °C^{-1}
ρ = density of water, 1,000 kg/m^3

The stratification conditions were found for two different models and the data, to separate as follows:

$S > 10^8$	no stratification
$10^8 > S > 10^7$	diurnally mixed
$10^7 > S$	always stratified

Condie and Webster (2001) also present an argument that mixing caused by flow through is negligible compared to that caused by even light winds. For conditions of operation of FWS treatment wetlands, these criteria predict no stratification.

The presence of vegetation promotes turbulence induced by water flow, but suppresses mixing caused by wind shear. Emergent vegetation canopies intercept a significant fraction of incident radiation, and thus prevent heating of the top

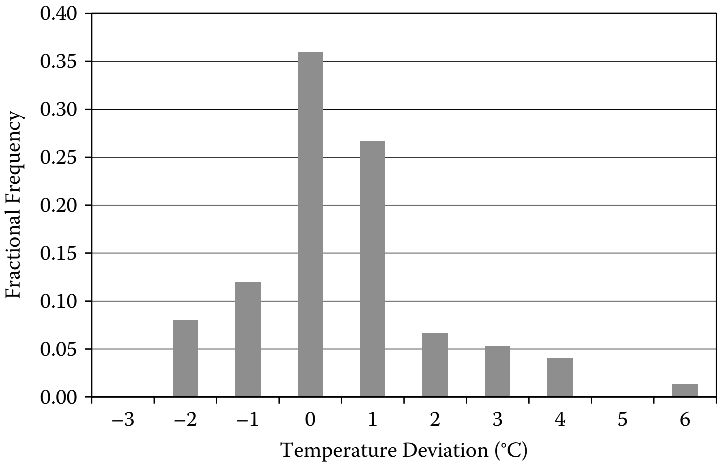


FIGURE 4.16 Deviations of weekly temperatures from the sinusoidal trend for the Grand Lake, Minnesota, HSSF treatment wetland.

layer of water. Therefore, the most extreme case would be expected for submerged aquatic vegetation (SAV), which can efficiently intercept radiation within the top layer of the water column, due to submerged leaves, yet inhibit wind and flow induced mixing. That is indeed the case for wetlands studied by Chimney *et al.* (2006). The surface of SAV beds was about 2.5°C warmer than water at 40–60 cm depth, based on average profiles over 18 months of the study. In contrast, surface temperatures and those at 40–60 cm depth differed by less than 0.5°C in *Typha* beds.

In HSSF wetlands, vertical stratification is inhibited by the thermal inertia of the wetland bed media. Further, solar radiation does not impinge directly on the water body, but is intercepted by the canopy and top layer of the gravel. As a consequence, stratification is minimal.

In general, temperatures in both FWS and HSSF wetlands are nearly uniform vertically. Although slight thermal stratification does exist in these treatment wetlands, the degree of temperature differential is usually small, and the top-to-bottom variation is typically not more than 1°C (Table 4.8).

In VF wetlands, the flow direction is perpendicular (normal) to vertical stratification mechanisms. The water column experiences a significant fraction of the cyclical soil temperature profiles that produce the dominant heat flux during the cold season. Vertical temperature gradients are not large (Table 4.8). Results from pilot scale VF wetlands indicate that the annual water temperature cycle is not much different from those for HSSF and FWS wetlands. The outlet water temperature is sinusoidal, with a 2°C winter minimum (Figure 4.17). Energy balance models for VF wetlands have been presented by Smith *et al.* (1997).

The Accommodation Zone

The inlet zone of a treatment wetland exhibits temperature changes, as the water approaches the balance temperature

TABLE 4.8
Vertical Temperature Profiles in Treatment Wetlands

HSSF Systems	Bed Depth (cm)	Bottom (cm)	Mid (cm)	Top (cm)
Grand Lake, Minnesota	60	53	23	8
Winter	T,°C	5.0	4.9	5.9
Summer	T,°C	16.5	17.9	21.8
NERCC, Minnesota	45	40	23	
Spring	T,°C	5.9	5.9	—
Summer	T,°C	16.2	16.1	—
Fall	T,°C	7.6	8.4	—
Minoa, New York	84	70	40	10
Planted				
Winter	T,°C	2.7	2.5	2.0
Spring	T,°C	8.2	8.3	8.9
Summer	T,°C	19.3	19.4	20.3
Fall	T,°C	17.7	17.7	17.7
Unplanted				
Winter	T,°C	5.0	4.9	4.4
Spring	T,°C	8.1	8.1	8.2
Summer	T,°C	20.3	20.1	20.1
Fall	T,°C	12.4	12.3	12.3
FWS Systems	**Water Depth (cm)**	**Bottom (cm)**	**Mid (cm)**	**Top (cm)**
ENR, Florida				
July— Cattail	70	60	30	20
	T,°C	28.43	28.29	28.41
July—Open Water	70	60	40	20
	T,°C	29.55	29.67	29.66
October—Open Water	70	60	40	20
	T,°C	24.94	25.08	25.13
VF Systems	**Water Depth (cm)**	**Bottom (cm)**	**Mid (cm)**	**Top (cm)**
Niagara-On-The-Lake, Ontario				
March	90	90	30	0
	T,°C	3.0	1.5	1.0

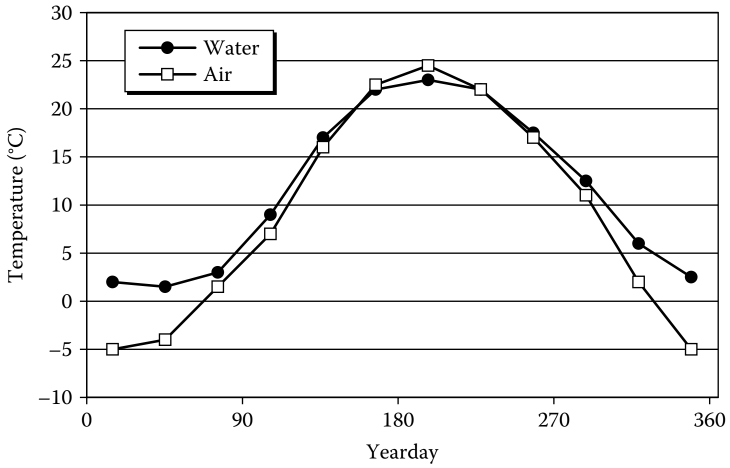


FIGURE 4.17 Annual progression of temperatures at the Niagara-on-the-Lake, Ontario, VF wetland. The measurement point was at 60-cm depth in a 90-cm downward flow path. The wetland was flood-dosed six times per day, totaling 6.0 cm/d. (Data from Lemon *et al.* (1996) *SWAMP pilot scale wetlands: Design and performance, Niagara-on-the-Lake, Ontario, Canada*. Presented at Constructed Wetlands in Cold Climates: Design, Operation, Performance Symposium; The Friends of St. George: Ontario, Canada.)

(see Figure 4.8). For short detention times (typically less than three days for FWS, and less than one day for HSSF), the adjustment may not be completed, and the balance temperature is not reached (Kadlec, 2006c).

Although the energy budget procedure (see Equation 4.1) is capable of providing a good representation of temperature variation with detention time, it is awkward to use because of the requirement for extensive meteorological data. Further, the partition between transpiration and evaporation is not known *a priori*, and it is only the evaporation component that affects water temperature; transpiration affects canopy temperature. An empirical exponential model may be easily calibrated and used to describe the approach to the balance temperature:

$$T_w = T_b + (T_{wi} - T_b)\exp\left(-\frac{\eta t}{\rho c_p h}\right) = T_b + (T_{wi} - T_b)\exp\left(-\frac{t}{\tau_A}\right) \quad (4.40)$$

where

c_p = heat capacity of water, 4.182×10^6 J/kg·°C
h = water depth, m
T_w = wetland water temperature, °C
T_b = wetland balance temperature, °C
T_{wi} = inlet water temperature, °C
η = accommodation coefficient, MJ/m²·d·°C
ρc_p = volumetric heat capacity of water, MJ/m³·°C
t = nominal detention time to an internal point, d

The quantity $\tau_A = \rho c_p h/\eta$ represents characteristic accommodation time for the wetland water on its travel through the system, during which 63.2% of the change from inlet to balance temperature has been achieved. At $3\tau_A$, 95.0% of the change has been accomplished.

The energy budget analysis suggests that the accommodation coefficient is comprised of radiative, evaporative, and convective components, with the radiative and evaporative portions being dominant (Kadlec and Knight, 1996). Therefore, although the accommodation coefficient is analogous to a convective heat transfer coefficient, and has the same units (MJ/m²·d·°C), it is not predictable from convection correlations as has been presumed in other literature (Reed *et al.*, 1995; Crites *et al.*, 2006), because those correlations ignore radiation, which is the principal heat input in summer, and soil heat retrieval, which is the major energy source in winter.

A further difficulty with previous wetland thermal literature is the reliance upon the assumption that the balance temperature is the air temperature, which is clearly not the case except in summer when the relative humidity is approximately 50%. It is further not the case in winter, when water temperatures are driven to within a degree or two of the freezing point, and not lower. A FWS wetland example illustrates this effect.

Warm-Up or Cool-Down?

The Tres Rios, Arizona, demonstration project operated 12 research wetlands (0.12 ha) and 4 pilot scale wetlands (about 1.0 ha). The research wetlands were operated at three detention times, approximately quadruplicated. Transects were monitored along the flow direction in the pilot wetlands. Consequently, on any given transect day, data were available for both distance and loading variations of detention time. Water temperatures coming from the advanced treatment plant were warm year-round, varying from 21–34°C.

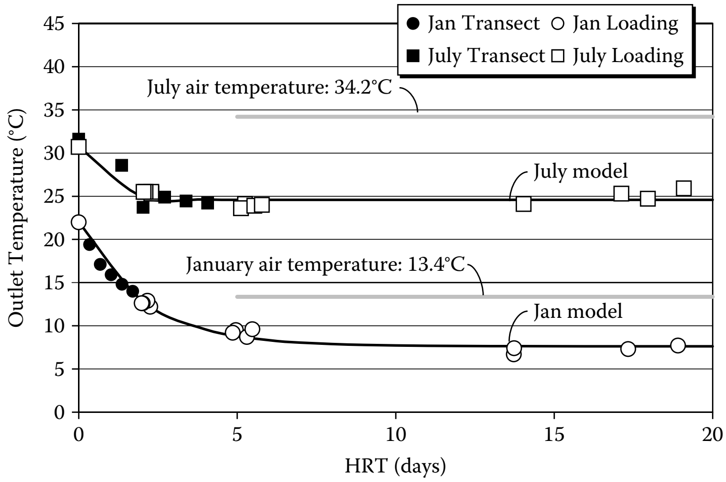


FIGURE 4.18 Wetland water temperature profiles through various Tres Rios, Arizona, FWS wetlands. Closed symbols represent a transect in wetland H1. Open symbols represent research wetlands operated at different detention times (loadings). The upper data and curves are for July 2; the bottom data and curves are for January 27.

The water cooled on passage through wetlands in both winter and summer (Figure 4.18).

Water temperatures display exponentially decreasing trends from the inlet water T_i to a balance temperature T_b. Balance temperatures were 5–10°C lower than the ambient air temperature, due to evaporative cooling in summer, and to evaporation and convection in winter. In summer, the Reed *et al.* (1995) convective model would suggest that the effluent at 31°C should warm up to the air temperature of 34°C, whereas operating data show that it cools to 25°C. An energy balance analysis (not shown) predicted a balance temperature of 26°C. In the summer, the relative humidity at the Tres Rios site is about 30%. Referring to Figure 4.14, it is seen that the corresponding prediction based upon ET_o (Equations 4.36 and 4.37) is 26°C.

This example represents an extreme of very hot arid conditions. Although there are no known temperature transect datasets for wet climates, it is to be expected that wetland balance temperatures would exceed air temperatures under such conditions. This is apparently true for the Hillsdale, New Hanover, and ENR datasets presented earlier, in Table 4.6, which all have long detention times, of about 20 days. Their effluent temperatures would then be balance temperatures. As further evidence of wetland water warm-up, Andradottir and Nepf (2000) found a 1–3°C temperature increase in littoral wetlands in the Boston area.

How Large Is the Adaptation Zone?

The wetland designer or data interpreter needs to know whether there is an adaptation zone, and if so, how much of the wetland it may occupy. This may be assessed either through estimates of η, the accommodation coefficient, or through τ_A, the time for 63.2% accommodation (see Equation 4.40). Data for FWS wetlands indicates that adaptation takes on the order of one to three days' detention (Table 4.9). This implies that many FWS treatment wetlands will totally contain the temperature adaptation gradient if the incoming water is colder or warmer than the balance point. As a result, very short detention wetlands may never reach the balance temperature, but most FWS systems will have an exit zone at the balance temperature.

The situation is different for HSSF wetlands, because of the thermal inertia of the media. Under arid conditions, for instance, evaporation has to cool the gravel as well as the water. Further, transpiration is probably more important than evaporation in HSSF systems than in FWS systems, as suggested by comparing Equation 4.24 to Equations 4.25 and 4.26. Nonetheless, HSSF water temperatures adapt during transit if there is a disparity between the incoming water temperature and the wetland balance temperature.

TABLE 4.9
Accommodation Coefficients (MJ/m²·d·°C) for FWS Wetlands and 63% Change Detention Times (τ_A) for Tres Rios, Arizona; Orlando, Florida, Easterly; and Sacramento, California, Wetlands

Wetland	Mean (η)	τ_A (days)	N
Tres Rios Research	0.97	1.47	240
Tres Rios Hayfield 1	0.57	1.80	23
Tres Rios Hayfield 2	0.62	1.67	22
Tres Rios Cobble 1	0.27	1.70	10
Tres Rios Cobble 2	0.43	1.69	11
Sacramento 3	2.50	0.78	2
Sacramento 5	1.33	0.98	3
Sacramento 7	1.45	2.11	4
Sacramento 9	0.65	3.70	2
Orlando Easterly	0.61	3.07	4

Note: N = number of transects or wetland months (research cells).

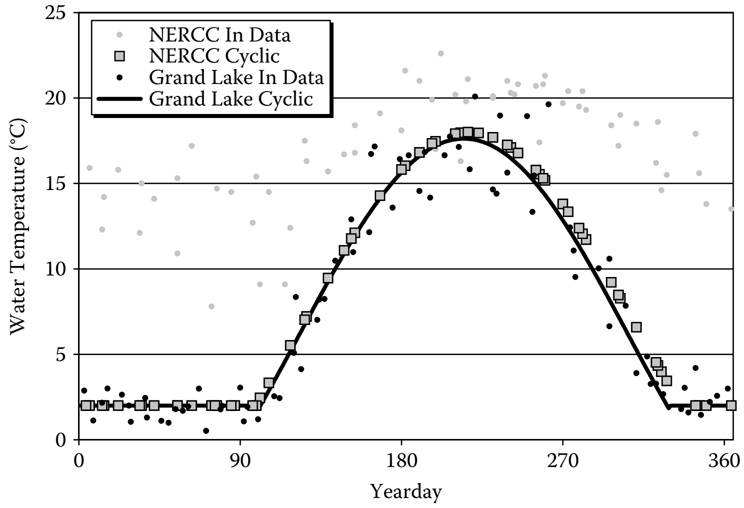


FIGURE 4.19 Annual temperature pattern for water in the Grand Lake and NERCC, Minnesota, HSSF wetlands. (From unpublished data; for more information, see Kadlec (2001b) *Water Science and Technology* 44 (11/12): 251–258.)

This was the case at the NERCC wetlands in Minnesota, which had warm water entering. The neighboring Grand Lake wetland received water at the local soil temperature. Both produced the same temperature effluents (Figure 4.19) due to similar energy flows. The NERCC HSSF wetlands had accommodation coefficients (η values) averaging 0.55–0.70 MJ/m²·d°C, corresponding to 95% adaptation in three to four days' detention. These values are similar to those for FWS systems (Table 4.9).

Longitudinal profiles were measured in the HSSF wetlands at Minoa, New York (Liebowitz *et al.*, 2000). In addition to measurements of temperatures at points along the flow path, three wetlands were operated in parallel at different hydraulic loading rates, hence different detention times. There is an exponential decline in temperature with nominal travel time (Figure 4.20). Cell 3 had short detention, and was entirely in the accommodation mode. Cells 1 and 2 had longer detention, and were mostly in the balance mode. Note that although the profile is for February, with an air temperature of about −4°C, the profiles trend to a balance temperature of +2°C.

Data for horizontal subsurface flow wetlands indicates that adaptation takes on the order of one day's detention (Table 4.10). This implies that many HSSF treatment wetlands will totally contain the temperature adaptation gradient if the incoming water is colder or warmer than the

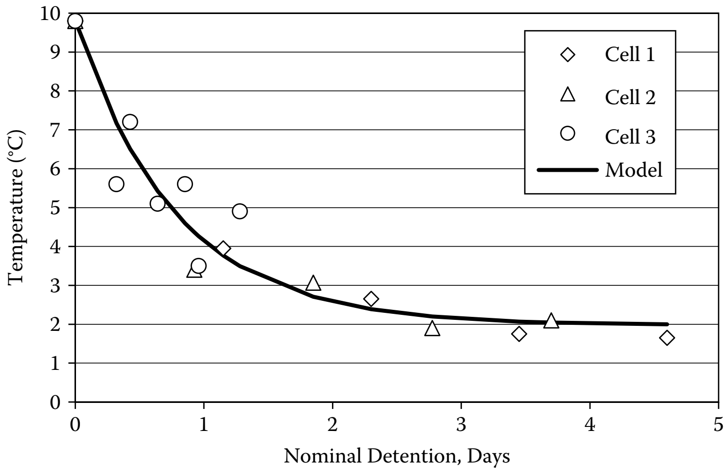


FIGURE 4.20 Temperature decrease in the flow direction through the three cells of Minoa, New York, HSSF wetlands on February 15, 1996. The three cells were operated in parallel at different detention times. Data points are averages for two depths, and one to three cross-flow positions. (Data from Liebowitz *et al.* (2000) *Subsurface flow wetland for wastewater treatment at Minoa, New York*. New York State Energy Research and Development Authority: New York.)

TABLE 4.10
Accommodation Coefficients (MJ/m²·d·°C) and 63% Change Detention Times (τ_A) for HSSF Wetlands

Wetland	Season	Balance T (°C)	τ_A (days)	Mean (η)	N
NERCC 1	Spring	7.7	1.86	1.0	21
	Summer	16.0	1.47	1.3	7
	Autumn	9.8	1.29	1.5	9
	Winter	1.7	1.03	1.8	20
NERCC 2	Spring	7.3	1.38	1.4	21
	Summer	15.7	1.69	1.1	6
	Autumn	10.1	0.82	2.3	9
	Winter	1.5	0.91	2.1	20
Minoa	Spring	7.1	0.82	3.9	1
	Summer	—	—	—	—
	Autumn	16.7	0.93	3.4	1
	Winter	2.0	0.78	4.1	1
Sacramento	Spring	—	—	—	—
	Summer	21.3	1.58	1.6	8
	Autumn	12.19	1.14	2.2	8
	Winter	10.81	0.39	6.5	3

balance point. As a result, most HSSF wetlands will operate over most of their length at the balance temperature.

4.4 COLD CLIMATES

Treatment wetlands that operate in cold (subfreezing) environments face several unique design challenges. During periods below freezing, the water temperature can no longer be approximated by air temperatures once an ice layer forms on the wetland. Effluent water temperatures will be 1 to 2°C, and the thickness of the ice layer becomes a design consideration. The formation of an ice layer will reduce the depth of the water column, reducing detention times, unless the water level is increased in the fall to accommodate the anticipated thickness of the ice layer. As a result, FWS wetlands in cold climates are often designed with additional freeboard in order to accommodate the anticipated layer of ice. Energy balance calculations are required to determine the extent of ice formation.

TABLE 4.11
Example of the Cumulative Effect of Insulation Layers for an HSSF Wetland

	Thickness (cm)	Thermal Conductivity (MJ/m·d·°C)	Resistance (MJ/m·d·°C⁻¹)
Air above/in canopy ($U = 0.3$)	—	—	3
Snow	25	0.010	25
Peat mulch	10	0.005	20
Dry gravel	5	0.026	2
Total	—	—	**50**

Ice thickness can vary significantly from year to year due to variations in snowfall and temperature. The principal factor is the insulation provided by the snow layer. Areas of emergent wetland vegetation are much more effective in trapping snow than unvegetated areas. Therefore, the thickness of ice in wetlands is much less than in adjacent lakes or frost depths in nearby uplands. Due to the spatial variability within the wetland, and year-to-year variations in winter conditions, simplifying assumptions are typically used to estimate ice formation.

The options that may be used for FWS treatment wetlands in cold climates include:

- Full year-round discharge, allowing for ice formation
- Restricted winter discharge accompanied by partial pond storage, and accelerated discharge through FWS treatment wetlands during the unfrozen season
- Storing water in ponds over the frozen season, and discharge through FWS treatment wetlands during the unfrozen season

These design options are explored in Chapter 17. HSSF wetlands provide further options, including:

- Added insulation, supported by the bed media or standing dead plants and thus kept out of the water. Mulch is one option (Wallace *et al.*, 2001), and is discussed in detail in this chapter. Straw may be used to supplement the standing dead plant material. Blankets, supported by the standing dead plant litter, have also been used.
- Lowered water levels, to create a layer of dry media (Jenssen *et al.*, 1994a).
- An ice layer on top of dry media. This is accomplished by raising water levels slightly above the media at the time of freeze-up. After the surface water freezes, the water level is dropped below the media surface, creating a dry media gap sealed by ice (Jenssen *et al.*, 1994a; Mæhlum, 1999).
- Using deep beds that allow for ice formation and retain capacity to pass water under the ice (Jenssen *et al.*, 1996).

In this section, methods for estimating the extent of ice formation are presented. Ice cover in wetlands causes the energy balance to split into a balance on the canopy and a separate balance on the water and ice below. It is the latter that is of interest in understanding the degree of ice formation. Radiation and vaporization are no longer factors for the water-side balance, because the ice layer blocks these processes from the underlying water.

FIGURE 4.21 The inlet zone of this FWS wetland at Hillsdale, Michigan, remains unfrozen in this February 1, 2001 photo. Downstream portions of the wetland are fully ice and snow covered. Note the preferential channel extending outward to the muskrat mound.

Spatial Extent and Distribution of Ice in FWS Wetlands

The point discharge introduction of warm water into either a constructed or natural treatment wetland causes an unfrozen, un-snow-covered inlet area to persist even in the event of extremely cold air temperatures. As the water moves out into the wetland, the incoming exotherm is dissipated, and a snow and ice cover becomes possible. This cover may consist of snow, ice, or a combination, depending on the vegetation density. If the discharge is into an unvegetated inlet zone, snow trapping is not possible, and ice covers the inlet pond in areas away from the discharge point. Flow then proceeds away under or over the ice. If the inlet zone is densely vegetated, snow may be held up above the water by standing dead vegetation or by a floating litter layer. In that event, flow from the unfrozen, uncovered inlet area proceeds away under a snow blanket.

When warm water enters a FWS wetland during freezing conditions, the incoming sensible heat may be sufficient to maintain an ice-free zone near the inlet, for at least part of the frozen season (Figure 4.21). At the Hillsdale, Michigan, wetlands, the unfrozen zone advances and retreats during freezing temperatures, depending upon short-term meteorological conditions. A complete ice cover may form under very cold conditions, only to have open water appear during less-cold, but freezing conditions. The incoming water is from venting groundwater, which has an annual temperature cycle from 4 to 21°C, with a mean of 12.5°C. It is this sensible heat that is capable of overcoming freezing for a short period of travel time.

This adaptation zone is not easily amenable to model calculations, because all of the terms in the energy balance (see Equation 4.1) are operative. A rough approximation may be obtained from Equation 4.40. An assumption of the accommodation coefficient must be made, based on Table 4.9, for instance. It is also necessary to assume that the balance temperature is approximately equal to the air temperature for the subfreezing conditions:

$$T = T_a + (T_i - T_a)\exp\left(-\frac{U}{\rho c_p v}y\right) \tag{4.41}$$

Equation 4.41 is solved for the distance at which the water temperature reaches 0°C. For instance, suppose $v = 4$ m/d, $T_a = T_b = -5$°C, $T_i = 5$°C, and $U = 1.0$ MJ/m²·d·°C. Then:

$$0 = -5 + (5 - (-5))\exp\left(-\frac{1}{(4.182)(4)}y\right)$$

$$y = 12 \text{ m}$$

These conditions approximate those for Figure 4.21, and correctly predict a small zone of open water at the inlet. Further down the flow path, for about 90% of the wetland, ice is present, and the energy balance changes markedly for under-ice conditions. The balance temperature increases to 1°C, because heat losses are reduced by the ice and snow cover.

The ability of warm incoming water to reverse ice formation has important consequences for system operation. At the NERCC, Minnesota, HSSF site, shutdown due to accidental damage caused the wetland water to freeze. But upon reinstatement of flow, the wetlands regained flow capacity, and returned to unfrozen operation. This self-healing capacity is fortunate, but may not be relied upon for routine operations.

The overall ice cover for the wetlands responds to the air temperature (Figure 4.22). When the mean air temperature drops below zero, ice formation quickly covers most of the wetland, excepting the inlet zone. In most years, this cover remains intact until the mean air temperature rises above freezing, whereupon the ice disappears rapidly, over a week or two.

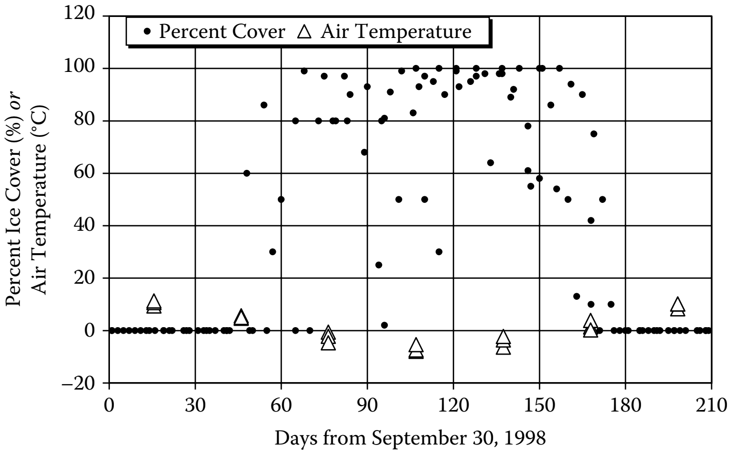


FIGURE 4.22 Extent of the ice cover at the Hillsdale, Michigan, FWS treatment wetlands. Data span four winters, for four wetland cells. The average monthly air temperature swing was from −4 to 22°C, with an annual mean of 9°C. Note that the proverbial "January thaw" was indeed experienced, with nearly complete loss of the ice cover during January.

Ice in Quiescent Ponds

An approximate estimating method for determination of ice thicknesses is based upon the length and intensity of subzero conditions. Ice formation in still water at a constant cold surface temperature proceeds according to the well-known Stefan model (Ingersoll *et al.*, 1948):

$$h_{\text{ice}} = a\sqrt{(-T_s \cdot t)} \tag{4.42}$$

where

a = constant, m/(°C·d)$^{0.5}$
h_{ice} = ice thickness, m
t = time, d
T_s = ice top surface temperature, °C

Crites *et al.* (2006) suggest that the value of a = 0.027 m/(°C·d)$^{0.5}$ for open water, 0.018 for open water with snow, and 0.010 for dense vegetation and litter; however, no calibration data or source are offered.

However, it must be noted that Equation 4.42 is for still, open waters, with no incoming sensible heat, and a fixed below-zero temperature at the top ice surface, none of which prevail in treatment wetland projects. Snow insulation, heat gain from soils, and moving wind and water all combine to slow ice formation. Consequently, the predicted values of ice thicknesses for wetlands are very conservative, often by factors of two or three. For instance, winter operation has been proven feasible at FWS projects located in Ontario, where the water would be predicted to freeze to the bottom according to Equation 4.42. Consequently, a more detailed energy balance is required for treatment wetland estimates.

The Balance Condition for Under-Ice Flow

The system chosen for energy balancing is selected to be the water under the ice, plus the wetted gravel matrix for the case of HSSF wetlands (Figure 4.23). Any storages of sensible heat in the ice layer, or in the wetland water body, are ignored. Any net heat loss is compensated by ice formation and cooling:

$$(H_a - G) = \left(\rho\lambda_f \frac{dh_{\text{ice}}}{dt}\right) \tag{4.43}$$

where

G = heat gain from deep soil, MJ/m^2·d
H_a = heat loss to air, MJ/m^2·d
h_{ice} = ice thickness, m
q = hydraulic loading, m/d
t = time, d
T = water temperature, °C
y = distance, m
λ_f = heat of fusion of water, MJ/kg
ρ = density of water, kg/m^3

The heat loss to air may be represented as a heat transfer coefficient times a temperature difference:

$$H_a = U(T - T_a) \tag{4.44}$$

where

T_a = air temperature, °C
U = overall heat transfer coefficient, MJ/m^2·d·°C

The balance condition prevails after the water has lost any excess sensible heat during its initial travel distance ($\partial T/\partial y = 0$), and has achieved a fixed temperature (T_b):

$$\lambda_f \frac{dh_i}{dt} = U(T_b - T_a) - G \tag{4.45}$$

The overall heat transfer coefficient (U) includes components due to air-side and water-side boundary layers, plus the ice

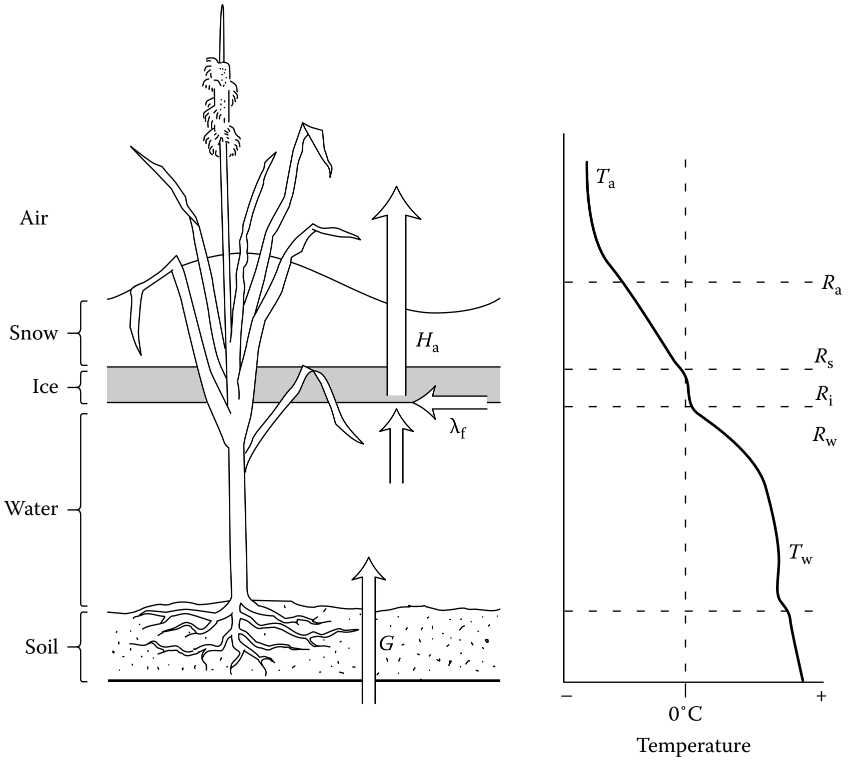


FIGURE 4.23 Temperature gradients through the vertical profile of a FWS wetland in winter. Partially frozen, balance conditions are assumed.

and snow layers, which greatly decrease the heat loss. The overall heat transfer coefficient is comprised of the several layer components:

$$\frac{1}{U} = \frac{1}{U_{water}} + \frac{h_{ice}}{k_{ice}} + \frac{h_s}{k_s} + \frac{h_m}{k_m} + \frac{1}{U_{air}} \tag{4.46}$$

where

h_{ice} = ice thickness, m
h_m = mulch thickness, m
h_s = snow thickness, m
k_{ice} = thermal conductivity of ice, MJ/m·d·°C
k_m = thermal conductivity of mulch, MJ/m·d·°C
k_s = thermal conductivity of snow, MJ/m·d·°C
U_{water} = water to ice heat transfer coefficient, MJ/m²·d·°C
U_{air} = snow to air heat transfer coefficient, MJ/m²·d·°C

The under-ice, water-side heat transfer coefficient has not been the subject of research. It is tempting to estimate the value based upon known relationships simple geometries. There are relationships for laminar and turbulent flow near flat plates, but these do not deal with the tortuous flow path and under-water obstructions (Welty *et al.*, 1983). There are also relations for arrays of pipes and packed beds, but these do not apply to the bounding walls of the enclosure (Welty *et al.*, 1983). Based on these other geometries, the water-side heat transfer coefficient is expected to be higher than the air-side coefficient for FWS wetlands.

The purpose of energy balance calculations for the frozen season is the estimation of either the amount of ice formed (FWS), or the amount of insulation needed to prevent ice formation (HSSF) (Wallace and Knight, 2006). The conservative assumption is that there is zero resistance to heat transfer on the water side of the air–water interface ($1/U_{water} = 0$), for both FWS and HSSF wetlands.

Example Detailed FWS Ice Calculation

Because of the time series progressions of air temperature and other variables, calculations require spreadsheet techniques for the solution of Equations 4.45 and 4.46, on a daily basis. There is an accommodation zone, in which incoming warm water cools to the winter balance point of 1–2°C. This example focuses upon energy fluxes in a downstream zone, in which that balance condition prevails. The driving force for ice formation is determined by the air temperature, which is presumed to follow a sinusoidal time progression fitted to data from 1997–1999 from Duluth, Minnesota. The annual swing in mean daily temperature is –11 to 19°C:

$$T_{air} = 4.2\left(1 + 3.57\cos\left[0.01721(t - 204)\right]\right) \tag{4.47}$$

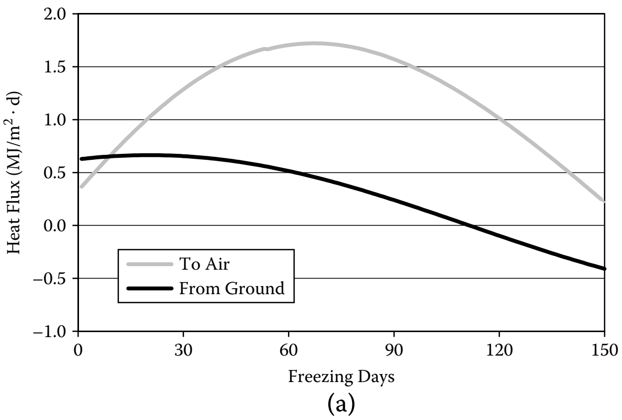
This leads to 150 days of freezing air temperatures, with a mean of –7°C, from November 8 to April 7. The water temperature in the balance zone is assumed to be 2°C for the frozen season. The heat loss transfer coefficients for the air-side of the ice is assumed to be 0.15 MJ/m^2·d·°C. For illustration, a snow-free condition is presumed (worst case). Consequently, the heat transfer resistance to loss to air is comprised of just three components: water-side, ice, and air-side.

Heat from the underlying ground tends to counterbalance heat loss to the atmosphere. For this example, this heat addition is assumed to be that of the cyclical model described above. Importantly, energy is returned from the soil to the water during most of the freezing period. The pattern is given by Equation 4.17:

$$G = \left[\frac{0.05(8.71)}{0.93}\right] \cdot \left(\cos\left[0.01721\left(t-195\right)\right] - \sin\left[0.01721\,(t-195)\right]\right) \tag{4.48}$$

The peak heat addition of 0.66 MJ/m^2·d occurs early in the freezing period, on November 30. After February 28, soil heat supply has been exhausted.

For these extreme conditions, 41 cm of ice are forecast (Figure 4.24). There were 1,058 degree-days below zero, resulting in a Stefan prediction of $(0.027 \times (1{,}058)^{0.5})(100) =$ 88 cm, which is much larger.



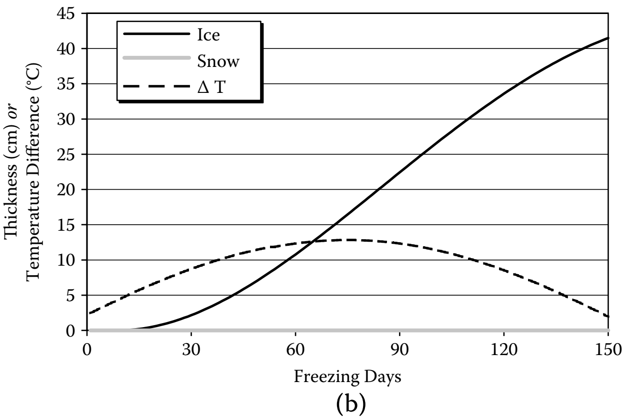


FIGURE 4.24 Progression of heat flows (a) and ice thickness (b) for an example FWS wetland with no snow. Note that the soil heat and warmer air temperatures slow ice formation during the early portion of the freezing period, and warmer air temperatures slow it during late winter.

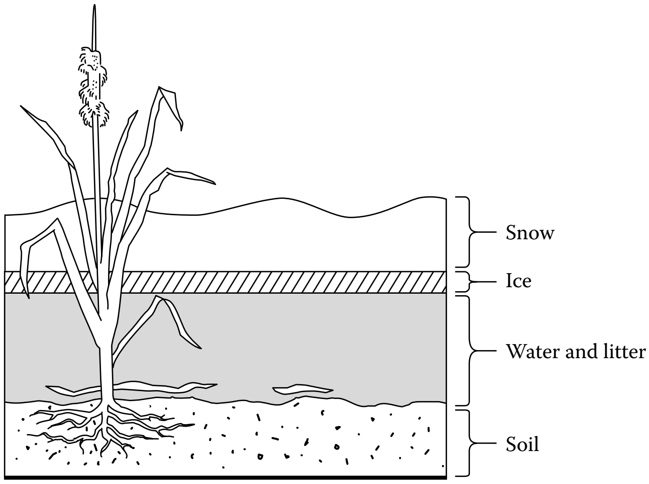


FIGURE 4.25 Cross section of a FWS wetland in winter.

A common occurrence is the collection of snow on top of the ice layer (Figure 4.25). If snow is considered, there is a considerably different result for the modeled ice thickness. A presumed pattern of snow accumulation over the first 20 days, persisting until March at 15 cm depth, then melting over 20 days, produces only 16 cm of ice. Further, the start of ice formation is delayed until mid-December. We note that this early winter phenomenon of snow in the canopy and unfrozen water beneath is common in cold temperate wetlands (see Figure 4.26).

Insulation of HSSF Wetlands

Because HSSF systems can be insulated by the addition of dry gravel and mulch layers, the balance condition energy fluxes can be modified to prevent ice formation (Henneck *et al.*, 2001; Wallace *et al.*, 2001; Kadlec, 2001b; Wallace and Knight, 2006). These layers add heat flow resistance over and above that which occurs naturally in the wetland, (which include standing dead, litter, and the snow trapped in the senesced vegetation). These natural insulation effects can be very important (see Figures 4.27 and 4.28), and may in fact be one of the most important thermal functions of the vegetation during the winter months.

Equation 4.46 provides the basis for evaluating insulation requirements. It is more intuitive to deal with heat flow resistances R ($R = 1/U$ or $R = h/k$), which are normally used in the insulation business. Basically, there is a need to estimate the total "R factor" needed, and to then calculate the amount of this that must be supplied in the form of mulch or dry gravel layers. As indicated in Equation 4.46, contributions are provided by the air above the wetland canopy, the litter and snow layer, mulch, and dry gravel.

The only heat source is the return flux from deep soils underneath the wetland, which varies through the winter as

FIGURE 4.26 Winter conditions at a constructed FWS wetland near Hillsdale, Michigan. Note the snow that is held up out of the water by standing dead and litter.

described by Equation 4.17. Heat loss to the atmosphere is driven by the mean daily temperature difference between the water (typically about 2°C) and the air (typically ranges from 0°C to −10°C). It is possible to allow the balance temperature to drop to zero during this period. The "bottleneck" for heat loss therefore typically occurs late in winter, when soil heat return is minimal, but the air temperatures are still subzero. The month of March brings increases in solar radiation, which raise the snow surface temperature, and thus decrease the driving force for freezing. The focus is then on the month of February, during which the mean energy flows are set by:

$$\frac{(T_b - T_a)}{R} = G - (\rho c V)\frac{dT_w}{dt} \tag{4.49}$$

where

c = heat capacity of water plus gravel, MJ/kg·°C
G = heat gain from deep soil, MJ/m^2·d
V = volume of water plus gravel in bed, m^3/m^2
R = overall heat transfer resistance, MJ/m^2·d·°C
t = time, d
T_w = water temperature,°C
T_a = air temperature,°C
T_b = water balance temperature,°C
ρ = density of water plus gravel, kg/m^3

The ground heat flux in February was approximately 0.125 MJ/m^2·d at the Grand Lake and NERCC sites. The air temperature for those sites averages −9.4°C in February. Allowable cooling of the wetland releases some heat. The

FIGURE 4.27 Winter conditions at a constructed HSSF wetland near Duluth, Minnesota.

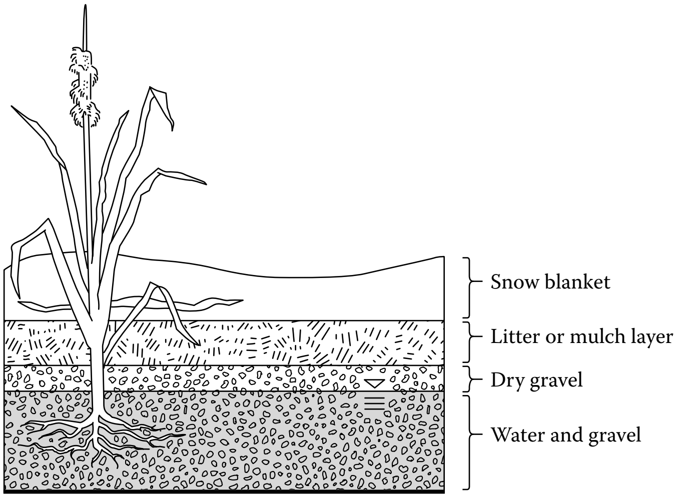


FIGURE 4.28 Cross section of a SSF wetland in winter.

heat capacity of the HSSF wetland is estimated to be 1.32 MJ/m^2·°C (see Equation 4.22). An allowed temperature decrease of 2°C per 30 days can therefore absorb (2 × 1.32/30) = 0.088 MJ/m^2·d.

The allowable balance temperature averages 1°C (2 down to zero). Therefore:

$$\frac{(1-(-9.4))}{R} = 0.125 - (-0.088) \tag{4.50}$$

Therefore, the necessary R = 49 (MJ/m^2·d·°C)$^{-1}$. It is comprised of several contributions, for instance (see Table 4.2 for thermal conductivities):

The expected snow depth for February for Duluth (and other locations in the United States) may be found from information at the National Climatological Data Center (NCDC), website:

http://cdo.ncdc.noaa.gov/climatenormals/clim20/

For the period 1971–2000, February had 22 days with greater than 25 cm of snow on the ground, and 27 days with greater than 12.5 cm. That much snow, of intermediate age, provides only about half the necessary insulation. The presence of a dry gravel layer is of little use. The air-side resistance is low, and also contributes little to reducing heat loss. The reader may verify that a layer of ice on top of the gravel is similarly of little or no use.

In the winter climate of Minnesota, 15 cm of mulch insulation has generally been sufficient enough to insulate the wetland bed and keep it from freezing, even during cold snaps of −45°C.

Warm Water Influents to HSSF Wetlands

As for FWS systems, when warm water enters a HSSF wetland during freezing conditions, the incoming sensible heat can maintain an ice free zone near the inlet, for some portion of the travel distance. Data on the rate of temperature decline with distance may be used to estimate the flow length over which freezing is prevented by incoming heat. The number of HSSF wetlands for which such temperature profiles are known is small. This is because of lack of monitoring, but also because waters entering HSSF wetlands often come from cool sources such as underground transfer lines, and thus enter at temperatures comparable to the balance temperature.

The question may be posed as to whether the warm incoming water will prevent freezing, and if so, for what travel length or equivalent detention time. The empirical exponential decline is a convenient assumption:

$$T_w = T_b + (T_i - T_b)\exp\left(-\frac{t}{\tau_A}\right) \tag{4.51}$$

Calibrations of Equation 4.22 for HSSF wetlands are shown previously, in Table 4.10. An example of the temperature change with nominal detention time is shown in Figure 4.20. The mean accommodation time (63% of the change) for HSSF wetlands is 1.15 days' nominal detention. Therefore, 95% of the incoming exotherm is lost in about three days' nominal detention. Three days' detention is the 90th percentile of the distribution for HSSF wetlands in the United States (N = 65 wetland-years). The median HRT for 28 HSSF systems in New South Wales, Australia, was 8.3 days (Davison *et al.*, 2005). Czech HSSF wetlands have median nominal detention time on the order of four days. Thus, most HSSF wetlands will not be prevented from freezing by incoming warm water. The balance temperature will be controlling in the bed outlet region, which is in turn controlled by soil heat and losses to the atmosphere.

SUMMARY

Many treatment wetlands exhibit a strong "buffer" capacity with respect to temperature due to energy flows within the wetland. Solar radiation is the driving force for evapotranspiration, which displays a strong annual cycle. In moderate temperate climates, ET losses are on the order of half a meter per year, but can easily be double that in hot arid climates.

Wetland exit water temperatures are approximately equal to the mean daily air temperature during unfrozen seasons, for conditions of moderate humidity and air temperature. Hot dry conditions can produce cooling, whereas humid conditions can produce heating. The equilibrium wetland water temperature represents a balance between the dominant transfers, which are incoming solar energy gains (R_N) and evaporative energy losses ($\rho\lambda_m ET$). The adjustment of the incoming water temperature to this balance occurs at a modest pace, with acclimation complete in about five days' detention. In the winter, the insulation provided by snow, ice, and mulch is enough to prevent water from freezing under cold climate conditions. The ice thickness is then determined by losses upward through the insulating layers as well as gains vertically upward from the earth by conduction. Energy balance equations permit calculation of wetland water temperatures.

5 Air, Water, and Soil Chemical Interactions

The physical and chemical environment of a wetland affects all biological processes. In turn, many wetland biological processes modify this physical/chemical environment. Four of the most widely fluctuating and important abiotic factors are dissolved oxygen (DO), oxidation-reduction potential (ORP), hydrogen ion concentration (pH), and alkalinity. Oxygen is frequently an influential factor for the growth of plants and animals in wetlands. Wetland plants have physiological adaptations that allow growth in low oxygen soils. Nitrification and oxidative consumption of organic compounds and BOD are dependent on dissolved oxygen. Wetland soils almost invariably are devoid of free oxygen, but still support a wide variety of oxidation and reduction reactions, such as ferric–ferrous iron conversion. The chemistry and biochemistry within the soil column are strongly driven by ORP. Hydrogen ion concentration, measured as pH, influences many biochemical transformations. It influences the partitioning of ionized and unionized forms of carbonates and ammonia, and controls the solubility of gases, such as ammonia, and solids, such as calcite. Hydrogen ions are active in cation exchange processes with wetland sediments and soils, and determine the extent of metal binding. Dissolved carbon dioxide, a major component of alkalinity, is the carbon source for autotrophic microbes and is the fundamental building block of wetland vegetation.

These variables may be understood by examining the normal ranges of variation in treatment wetlands. Successful design also requires that forecasts be made for intended operating conditions, which in turn implies prediction rules and equations.

It has been suggested that wetland plants are merely the substrate for microbes, which function as they would in a trickling filter. Indeed, some have suggested that the plants can be replaced by wooden or plastic dowels at the same stem density. Nothing could be further from the truth. Wetland plants are actively passing gases, both into and out of the wetland substrate. The more correct image is of a forest of chimneys, sending plumes of various gases into the atmosphere, interspersed with other plants acting as air intakes. On the diurnal cycle, the entire wetland "breathes" in and out, bringing in oxygen and discharging carbon dioxide, methane, and other gases.

5.1 FUNDAMENTALS OF TRANSFER

A FWS wetland provides considerable opportunity for losses of volatile compounds from the water to the atmosphere, and transfers of oxygen and carbon dioxide from the atmosphere, as does a VF system. However, HSSF wetlands have restricted ability to accomplish those transfers, because of the presence of the bed media and possibly mulch. The large areal extent, coupled with relatively long detention times and shallow water depths, are conditions that foster convective and diffusional transport to the air–water interface, upward to bulk air, and laterally off-site under the influence of winds (Figure 5.1). There is typically equilibrium between air-phase and water-phase concentrations at the interface, which separates two vertical transport zones.

Henry's law expresses the equilibrium ratio of the air-phase concentration to the water-phase concentration of a given soluble chemical. A variety of concentration measures may be used in both phases, thus generating several definitions of Henry's Law Constant (H). Here the water phase concentration is presumed to be given as mmol/L = mol/m^3, and the gas phase concentration as partial pressure in Pascals (Pa) (mole or volume fraction times total pressure). Thus:

$$P_{\text{interface}} = HC_{\text{interface}} \tag{5.1}$$

where

$C_{\text{interface}}$ = interfacial water phase concentration, mol/m^3
H = Henry's Law Constant, atm·m^3/mol
$P_{\text{interface}}$ = interfacial partial pressure in air, atm

Transport in both the air and water phases may involve convective currents as well as molecular diffusion, and therefore the transport flux (flow per unit area) is commonly modeled with mass transfer coefficients (Welty *et al*., 1983):

$$J - k_{\text{w}}(C - C_{\text{interface}}) = k_{\text{a}}(P_{\text{interface}} - P) \tag{5.2}$$

where

C = water phase concentration, mol/m^3
J = loss flux, mol/m^2·hr
k_{a} = air-side mass transfer coefficient, (m/hr)(mol/m^3)/atm = mol/(m^2·atm·hr)
k_{w} = water-side mass transfer coefficient, m/hr
P = partial pressure in air, atm

It is common practice to eliminate the unknown interfacial concentrations between Equations 5.1 and 5.2, yielding an

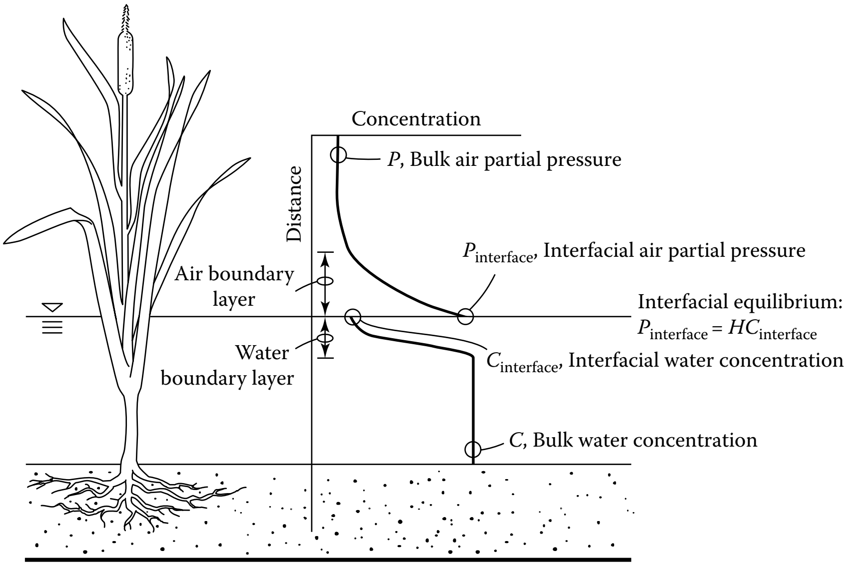


FIGURE 5.1 A soluble volatile chemical can move from the bulk water to the air–water interface, where it equilibrates with the air-phase chemical. Movement then occurs in the air, away from the interface out to the bulk air. These routes are reversed for chemicals being taken up. Transport is typically in the turbulent range in the air, and in the laminar or transition range in the water.

expression for transfer from the bulk water to the bulk air:

$$J = K_w\left(C - \frac{P}{H}\right) \tag{5.3}$$

$$\frac{1}{K_w} = \frac{1}{k_w} + \frac{1}{Hk_a} \tag{5.4}$$

where
K_w = overall water-side mass transfer coefficient, m/hr

In many instances of pollutant transfer, there is a zero bulk air concentration, and the transfer model reduces to:

$$J = K_w C \tag{5.5}$$

Air-side mass transfer coefficients are quite large, which places nearly all the mass transfer resistance on the liquid side. For instance, Mackay and Leinonen (1975) found over 80% of the transfer resistance in the water when $H > 10^{-4}$ atm·m^3/mol. It is again noteworthy that this theory leads to a first-order areal removal rate.

Values of k_w depend upon the degree of convective mixing, as well as on the size of the molecule being transported. A large body of knowledge concerning oxygen and other gases in ponds was reviewed by Ro *et al.* (2006) and Ro and Hunt (2006). They determined a general correlation from data concerning several gases:

$$K_w = 1.706\, Sc^{-0.5} U_{10}^{1.81} \left(\frac{\rho_a}{\rho_w}\right)^{0.5} \tag{5.6}$$

where
Sc = Schmidt number, = ν/D, dimensionless
D = diffusivity of gas, m^2/s
U_{10} = wind speed at 10 m height, m/s
ρ_a = density of air, kg/m^3
ρ_w = density of water vapor, kg/m^3
ν = kinematic viscosity of gas, m^2/s

Experimental studies of Peng *et al.* (1995) verified the strong effect of mixing in the water phase, and established a diffusion-only value of $k_w \approx 0.03$ m/h for benzene, toluene, TCE, and PCE. In the context of treatment wetlands, these rate constants are in the range of 20–2,000 m/yr. Therefore, light molecules are very likely to be effectively stripped in wetlands that are designed to remove other constituents with equal or lower rate constants.

Plants participate in the transfer of gases to and from air, via their internal airways. For oxygen, this transfer is called the plant aeration flux, and is required to support respiration and to protect the root zone. Because any excess oxygen is available in the root zone for processes such as nitrification, further discussion of this process is to be found in Chapter 9.

5.2 OXYGEN DYNAMICS IN TREATMENT WETLANDS

Dissolved oxygen (DO) is of interest in treatment wetlands for two principal reasons: it is an important participant in some pollutant removal mechanisms, and it is a regulatory parameter for discharges to surface waters. In the first instance, DO is the driver for nitrification and for aerobic decomposition of CBOD. In the second instance, DO is critical for the survival of fish and other aquatic organisms, and for the general health of receiving water bodies. In many permits in the United States, a minimum DO of 5 mg/L is specified.

Water entering the treatment wetland has carbonaceous and nitrogenous oxygen demand (NOD). After entering the wetland, several competing processes affect the concentrations of oxygen, biochemical oxygen demand (BOD), and nitrogen species. Dissolved oxygen is depleted to meet wetland oxygen requirements in four major categories: sediment/litter oxygen demand, respiration requirements, dissolved carbonaceous BOD, and dissolved NOD. The sediment oxygen demand is the result of decomposing detritus generated by carbon fixation in the wetland, as well as decomposition of accumulated organic solids which entered with the water. The NOD is exerted primarily by ammonium nitrogen; but ammonium may be supplemented by the mineralization of dissolved organic nitrogen. Decomposition processes in the wetland also contribute to NOD and BOD. Microorganisms, primarily attached to solid, emersed surfaces, mediate the reactions between DO and the oxygen consuming chemicals. Plants and animals within the wetland require oxygen for respiration. In the aquatic environment, this effect is seen as the nighttime disappearance of dissolved oxygen. Oxygen transfers from air, and generation within the wetland, supplements any residual DO that may have been present in the incoming water. Three routes have been documented for transfer from air: direct mass transfer to the water surface, convective transport down dead stems and leaves, and convective transport down live stems and leaves. The latter two combine to form the plant aeration flux, (PAF). These transfers are largely balanced by root respiration, but may contribute to other oxidative processes in the root zone.

Despite this complexity, wetlands are not particularly efficient at obtaining oxygen in sufficient quantities to deal with heavy pollutant loads. Therefore, several techniques have been employed to supplement the natural aeration processes. Compressed air bubblers, alternating fill and draw, and intermittent vertical flow have all been successfully implemented. These systems are described in more detail in Part II of this book; in this section the focus is upon passive treatment wetlands.

Biochemical Production of Oxygen

Oxygen is the byproduct of photosynthesis (Equation 5.7). When photosynthesis takes place below the water surface, as in the case of periphyton and plankton, oxygen is added to the water internally. A large algal bloom can raise oxygen levels to 15–20 mg/L, more than double the saturation solubility, as a result of wastewater addition (Schwegler, 1978). This process requires sunlight, and algal photosynthesis is suppressed in wetlands with dense covers of emergent macrophytes.

$$6CO_2 + 12H_2O + \text{light} \rightarrow C_6H_{12}O_6 + 6O_2 + 6H_2O \quad (5.7)$$

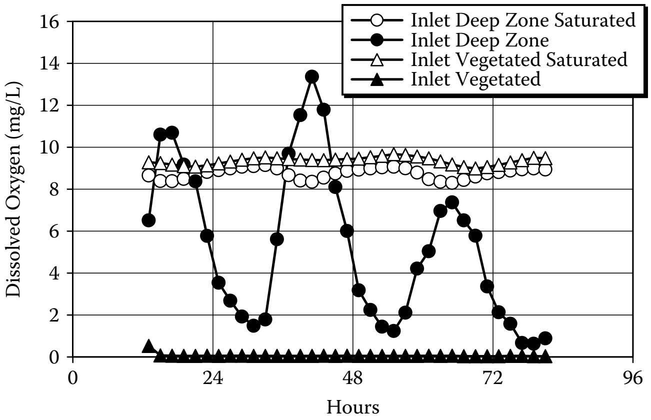


FIGURE 5.2 Diurnal cycles in dissolved oxygen in Cell 7 of the Sacramento, California, FWS treatment wetland project, May 28–31, 1996. The inlet deep zone exceeds saturation in late afternoon. Just 46 meters downstream, in a dense community of cattails and bulrush, there is essentially no dissolved oxygen, despite a slightly higher saturation value (the water has cooled slightly). (Data from Nolte and Associates (1998a) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project*. 1997 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates.)

Nonshaded aquatic microenvironments within the wetland therefore display a large diurnal swing in dissolved oxygen due to the photosynthesis–respiration cycle. Nutrients stimulate the algal community, and increase the DO mean and amplitude. When large amounts of nutrients are added to the wetland, and water depths are shallow enough for emergent rooted plants, other components of the carbon cycle are increased, such as photosynthesis by macrophytes. It is then possible for other wetland processes to become dominant in the control of dissolved oxygen. The effect is typically a depression of average DO, and a decrease in the amplitude of the diurnal cycle (Figure 5.2). This suppression of the diurnal DO cycle is a characteristic of all treatment wetlands receiving moderate to high loads of carbonaceous and nitrogenous oxygen demand.

In wetlands dominated by macrophytes, oxygen processing is more complicated. Macrophytes and periphyton contribute to respiration and photosynthesis. The decomposition of litter and microdetritus returns ammonium nitrogen and BOD to the water and to the root zone. Oxygen transfer to the root zone occurs through plants as well as from mass transfer. BOD can also degrade via anaerobic processes in the wetland litter and soil horizons.

Physical Oxygen Transfers

The concentration of dissolved oxygen (DO) in water varies with temperature, dissolved salts, and biological activity. The effect of temperature on the equilibrium solubility of oxygen in pure water exposed to air has been widely studied, and can be calculated from regression presented in Equation 5.8

(Elmore and Hayes, 1960):

$$C_{DO}^{sat} = 14.652 - 0.41022\,T + 0.007991\,T^2 - 0.00007777\,T^3 \quad (5.8)$$

where
C_{DO}^{sat} = equilibrium DO concentration at 1.0 atmosphere, mg/L
T = water temperature, °C

This relation shows that at 25°C, the equilibrium DO = 8.2 mg/L, while at 5°C, the equilibrium DO = 12.8 mg/L.

There are few studies of reaeration in wetlands, and therefore the rate of oxygen supply from the atmosphere can only be estimated. Here, the methods of quantification from stream reaeration are adopted. The applicable mass transfer equation is presented in Equation 5.9:

$$J_{O_2} = K_L\left(C_{DO}^{sat} - C_{DO}\right) \quad (5.9)$$

where
C_{DO}^{sat} = saturation DO concentration at water surface, mg/L = g/m^3
C_{DO} = DO concentration in the bulk of the water, mg/L = g/m^3
K_L = mass transfer coefficient, m/d
J_{O_2} = oxygen flux from air to water, g/m^2·d

The parameter K_L has been the subject of dozens of research studies in lakes and streams, and in shallow laboratory flume studies (U.S. EPA, 1985b). Four factors are important in determination of K_L: the velocity and depth of the water, the speed of the wind, and rainfall intensity.

The first two factors are typically dominant in streams and rivers, in which flow is turbulent. Accordingly, several equations in the literature are based on turbulent flow conditions, which typically do not prevail in FWS wetlands (see Chapter 2). Leu *et al*. (1997) have examined six such formulations, including the popular O'Connor and Dobbins (1958) correlation, in the context of data in laminar flow. The O'Connor and Dobbins (1958) correlation was found to greatly overpredict the mass transfer coefficient in low velocity situations (Leu *et al*., 1997).

More serious is the failure of many equations, including O'Connor and Dobbins (1958), to account for the extremely important effect of wind mixing. Chiu and Jirka (2003) present data from a large unvegetated mesocosm (1 m wide by 20 m long) that demonstrate an essentially direct proportionality between K_L and the square of the wind speed. In a FWS environment, the presence of vegetation blocks wind mixing preferentially for low wind speeds. Belanger and Korzun (1990), working in sparse *Cladium* and moderately dense *Typha* wetlands, found no effect of wind up to about 3.2 m/s (as measured at ten meter height), followed by a direct proportionality to the excess of wind speed above that threshold. Thus for light winds, up to 3.2 m/s, K_L = 0.2 m/d, whereas K_L increased dramatically to ten times that value at wind of 5.5 m/s. The presence of sparse emergent macrophytes therefore does not block physical oxygen transfer.

Low values of K_L in wetlands are due in large measure to low flow rates, and the attendant low degree of water mixing. In addition to the effect of wind, rain also creates surficial mixing and increases the mass transfer coefficient. Belanger and Korzun (1990) measured a linear dependence of K_L on rainfall intensity, with K_L = 1.2 m/d at a rainfall rate of 5 mm/h. Thermal convection, operating on a diurnal cycle, has also been implicated in oxygen transfer in treatment wetlands (Schmid *et al*., 2005a).

Open Water Zones

Treatment wetlands are sometimes configured with open water zones, which would seem to offer enhanced opportunity for oxygen transfer. Despite the considerable uncertainty in the mass transfer coefficient, calculations show that physical reaeration is a slow process, even under moderate windiness. For instance, in the absence of any other processes, the forecast of the detention time to bring water from zero DO to 90% of saturation is in the range of two to four days for typical wind velocities.

Bavor *et al*. (1988) operated an open water, unvegetated wetland receiving secondary effluent. This system maintained high DO, ranging from 4.3 to 14.6 mg/L over the seasons. The values of K_L calculated from Bavor's open water system were 0.2–1.0 m/d under some conditions. But oxygen levels frequently exceeded saturation, indicating internal generation of oxygen, most likely by algae. Suspended solids were quite high in the effluent, 24–147 mg/L.

An open water, unvegetated wetland was monitored for DO in Commerce Township, Michigan, for a period of three years. Ammonia and BOD were very low in this "polishing" wetland, typically less than 0.2 mg/L for ammonia and less than 2.0 mg/L for $CBOD_5$. Inlet DO averaged 83% of saturation, and outlet DO was 91% of saturation after 3.3 days' detention. The corresponding mean K_L value was 0.42 m/d (R.H. Kadlec, unpublished data).

The Tres Rios, Arizona, wetland H1 contained 20% deep zones (1.5 m) in seven sections, with 80% at a depth of 0.3 m. The deep zones were predominantly open water, with only occasional *Lemna* cover and sparse SAV. The incoming wastewater contained essentially no $CBOD_5$ (2.3 mg/L) and little ammonia (1.57 mg/L) during a three-year period in which DO profiles were measured. The mean detention time was 5.6 days. Wastewater entered at low DO, and was not oxygenated during transit (Figure 5.3). Thus, it appears that atmospheric reaeration of open water occurs only to a limited extent. No existing correlation for K_L can be recommended, because none have been developed for wetland conditions. As a preliminary estimate for FWS wetlands, $0.1 < K_L < 0.4$ m/d (R.H. Kadlec, unpublished data).

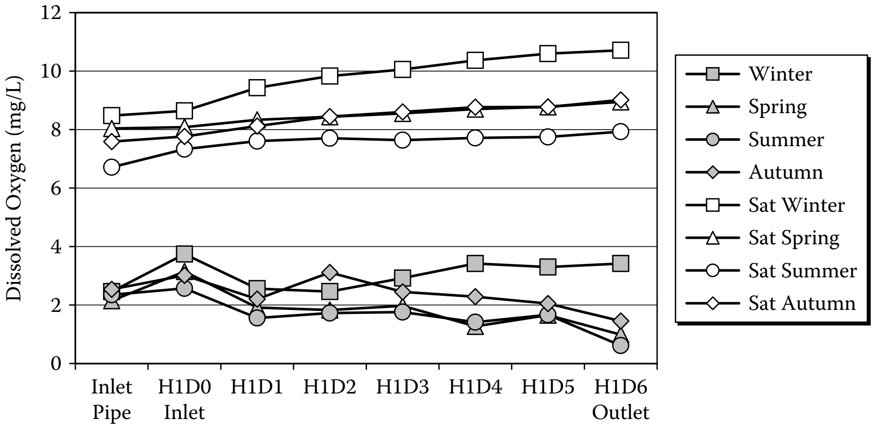


FIGURE 5.3 Dissolved oxygen profiles along the flow path through Hayfield Cell 1 at the Tres Rios, Arizona, site. Seasonal averages of monthly data collected over three years. The sampling points were located in deep zones located at even spacing from inlet to outlet. The detention time was 5.6 days, at a depth of 30 cm in the bench areas.

Plant Oxygen Transfer

Emergent Plant Oxygen Transfer

Great care must be exercised in the interpretation of the literature concerning oxygen transfer by plants in wetlands. Although it is certain that oxygen transfer does occur at modest rates, the amount that is transferred in excess of plant respiration requirements is much less certain. Further, methods of measurement have been variable, and some are purely presumptive. One group of estimates relies upon measurements for individual plants or roots, commonly in hydroponic environments, and extrapolation via root dimensions and numbers. For example, Lawson (1985) calculated a possible oxygen flux from roots of *Phragmites australis* up to 4.3 $g/m^2 \cdot d$, and Armstrong *et al.* (1990) calculated 5–12 $g/m^2 \cdot d$. Gries *et al.* (1990) calculated 1–2 $g/m^2 \cdot d$. It is apparent that the oxygen demand in the root environment is an important determinant of how much oxygen is supplied to that root zone, with high demands increasing the supply, up to a limit (Sorrell, 1999). Hydroponic systems react much differently to flow through than to batch conditions (Sorrell and Armstrong, 1994). Furthermore, plants growing in anoxic conditions can modify their root structure, creating fewer small roots and more large roots, presumably as a defense against the large oxygen supplies demanded by the small roots (Sorrell *et al.*, 2000). Nonetheless, such hydroponic experiments serve to elucidate the effects of variables. For example, Wu *et al.* (2000) used hydroponic experiments to estimate 0.04 $g/m^2 \cdot d$ supplied by *Typha latifolia*, versus 0.60 $g/m^2 \cdot d$ supplied by *Spartina pectinata*.

A second group of estimates relies upon the disappearance of CBOD and ammonia to infer an oxygen supply. Differences between side-by-side systems are then used to infer the amount of the inferred supply that came from plants. This procedure also has considerable uncertainty, because it is founded on the presumption of oxygen consumption being due to oxidative processes for ammonia and CBOD, and to specific stoichiometric relations. That presumed chemistry is in question, because of alternative loss and gain mechanisms for both ammonia and CBOD. Cooper (1999) labels the estimation of oxygen supply from ammonia and CBOD loss "a crude calculation." Consequently, such determinations are here termed "implied oxygen supply" rates. However, a number of authors have reported such implied oxygen supply (Platzer, 1999; Wu *et al.*, 2000; Crites *et al.*, 2006). Again, this estimate may be better used as a comparative, with reference to side-by-side studies of vegetated and unvegetated systems.

The third group of studies relies upon direct measurements of oxygen uptake. This may be done in the field (e.g., Brix, 1990), or more readily in laboratory mesocosms (e.g., Wu *et al.*, 2001). Brix (1990) and Brix and Schierup (1990) cast doubts upon the importance of oxygen release from plants, and more recent studies have confirmed this lack of importance. For instance, Townley (1996) found essentially no oxygen released by *Schoenoplectus* (*Scirpus*) *validus* or *Pontederia cordata*. Wu *et al.* (2001) measured 0.023 $g/m^2 \cdot d$ transferred by *Typha* in mesocosms. Bezbaruah and Zhang (2004; 2005) used direct measurement techniques to study the effects of BOD on oxygen transfer by *Scirpus validus*, and found only 1–4 $mg/m^2 \cdot d$ released at BOD = 76 mg/L, and 11 $mg/m^2 \cdot d$ released at BOD = 1,267 mg/L. This direct measurement evidence strongly suggests that emergent plants do not contribute "extra" oxygen transfer to any appreciable degree, although they do send oxygen to the root zone to protect themselves and conduct respiration. More information on oxygen transfer is presented in Chapter 9, in the context of nitrification.

Floating Plants

Open water zones, in the presence of elevated nutrient supplies, may be colonized by floating plants, such as *Lemna* spp., *Hydrocotyle umbellata*, and *Azolla* spp. These form a physical cover that is a barrier to oxygen transfer. Additionally, wind can cause the formation of very thick mats by drifting and compression. Root oxygen release rates from a number of free-floating plants in batch hydroponic laboratory studies were calculated in the range of 0.26–0.96 g/m²·d (Moorhead and Reddy, 1988; Perdomo *et al*., 1996; Soda *et al*., 2007).

As an example, the Sacramento, California, wetlands were configured with 19% of the area without emergent plants, due to design water depths of 1.5 m (Nolte and Associates, 1997). Most of the deep zones became covered with *Lemna* spp. On some occasions, DO concentrations increased in these deep zones, but on average there was little increase in DO. The ammonia loading was high, with concentrations in the range of 10–20 mg N/L. There was no discernible increase in the ammonia removal rates in the deep zones.

Submerged Plant Oxygen Transfer

Submerged aquatic vegetation (SAV), including algae, photosynthesizes within the water column, and therefore contribute oxygen directly to the water. This activity is driven by sunlight, leading to very strong diurnal cycles in the resultant DO content of the water column. The magnitude of DO enhancement can be large, especially in lightly loaded wetlands. Root oxygen release rates from a number of submerged plants in natural environments are reported to be in the range of 0.5 to 5.2 g/m²·d (Sand-Jensen *et al*., 1982; Kemp and Murray, 1986; Caffrey and Kemp, 1991). More recent work by Laskov *et al*. (2006) shows a calculated range of 0.15–0.60 g/m²·d based on 200 plants per square meter.

Attempts to relate the effect of oxygen transfer to ammonia removal, via the presumptive enhancement of added DO, are less than clear. For instance, the data of Toet (2003) details the performance of *Phragmites* and *Typha* in the first half of a FWS wetland, followed by submerged vegetation dominated by *Elodea nuttallii*, *Potamogeton* spp., and *Ceratophyllum demersum*. Eight wetlands plus an unvegetated control were studied for a calendar year, two years after startup. Organic loadings were very low, and ammonia was typically in the range 0.4 to 0.7 mg N/L. The emergent sections of the wetlands lowered the already-low DO from the pretreatment plant. The submergent sections raised the DO to 4–18 mg/L. However, ammonia removal rates were found to be lower in the submerged sections than in the emergent sections, with mass removal efficiencies more than two times lower (33% versus 12%).

DB Environmental (DBE, 2002) operated SAV mesocosms and 0.2 ha SAV wetlands during 1999–2002. Dissolved oxygen was found to be at or above saturation during the day in the surface water layer, but was very much lower at night and in bottom water layers.

Knight *et al*. (2003) reported the performance of 13 flow through Florida water bodies dominated by SAV. Of these, seven were in the depth range (1.1–2.2 m) and the detention time range (2–20 days) of interest for treatment wetlands. Incoming ammonia levels were low (0.03–0.20 mg/L), as were TKN levels (0.1–2.8 mg/L). These large systems (147–2,452 ha) removed no ammonia, and further did not alter TKN. Therefore, the implied oxygen supply was zero, thus casting more doubts on the use of ammonia removal as an indicator of oxygen supply in the SAV environment.

U.S. EPA (1999) shows high oxygen concentrations for the surface layer of the SAV sections of FWS wetlands operating in Arcata, California. However, the vegetative cover was not stable, changing from SAV to *Lemna* on a seasonal basis (U.S. EPA, 1999). U.S. EPA (2000a) hypothesizes the necessity for including a SAV zone in FWS design for ammonia removal, based upon presumptive reoxygenation. However, they state that "… quantitative estimates of transfer are difficult to assess based on current data."

Biological and Chemical Oxygen Consumption

Longitudinal Gradients

When wastewater with BOD and ammonia nitrogen is discharged to rivers and streams, an oxygen sag analysis is often applied (Metcalf and Eddy Inc., 1991). This Streeter–Phelps (1925) analysis is predicated on the assumption that oxygen is increased in the flow direction by mass transfer from the air above, and by photosynthesis occurring within the water column, and decreased by consumption of BOD and ammonium nitrogen oxidation, and decreased by consumption of Sediment Oxygen Demand (SOD) and respiration. In the wetland environment, both sediments and litter consume oxygen during decomposition. Decomposition processes also release carbon and nitrogen compounds to the overlying water, which can exert an oxygen demand. It is therefore apropos to designate the sum as Decomposition Oxygen Demand (DOD). Plants transfer oxygen to their root zone to satisfy respiratory requirements, and may in some instances transfer a surplus to control the oxygen environment around the roots. The balance on DO in the wetland from the inlet (0) to a specified distance (L) along the flow path can be written as (Equation 5.10):

$$\begin{aligned} q\left[C_{\text{DO}}(L) - C_{\text{DO}}(0)\right] &= K_L\left(C_{\text{DO}}^{\text{sat}} - \overline{C_{\text{DO}}}\right) \\ &+ \left[r_{\text{O, photo}} - r_{\text{O, res}} - r_{\text{O, DOD}}\right] + a_{\text{N}} q\left[C_{\text{N}}(L) - C_{\text{N}}(0)\right] \\ &+ a_{\text{B}} q\left[C_{\text{BOD}}(L) - C_{\text{BOD}}(0)\right] \end{aligned} \tag{5.10}$$

where

a_{N} = stoichiometric coefficient for NH_4-N oxygen demand
a_{B} = stoichiometric coefficient for BOD oxygen demand
$\overline{C_{\text{DO}}}$ = average DO concentration average over length L, g/m³ = mg/L
C_{BOD} = BOD concentration, g/m³ = mg/L
C_{N} = ammonia nitrogen concentration, g/m³ = mg/L
q = hydraulic loading rate, m/d
$r_{\text{O, photo}}$ = rate of DO generation by photosynthesis, g/m²·d
$r_{\text{O, res}}$ = rate of DO consumption by respiration, g/m²·d
$r_{\text{O, DOD}}$ = rate of DO consumption by decomposition, g/m²·d

There is no treatment wetland data with which to separately evaluate photosynthesis, respiration, plant aeration flux (PAF), and decomposition oxygen demand (DOD). It is necessary to lump these into Wetland Oxygen Demand (WOD) (Equation 5.11):

$$r_{O,\,WOD} = r_{O,\,DOD} + r_{O,\,res} - r_{O,\,photo} \qquad (5.11)$$

where

$r_{O,\,WOD}$ = net wetland oxygen consumption rate, g/m²·d

Further, there is often no data from which to estimate the reaeration coefficient K_L. Therefore, all transfer rates to and from the atmosphere and to and from the biomass in the wetland are lumped into a single term, the wetland net oxygen supply rate (Equation 5.12).

$$r_{NOSR} = K_L\left(C_{DO}^{sat} - \overline{C_{DO}}\right) - r_{WOD} \qquad (5.12)$$

where

r_{NOSR} = net oxygen supply rate, g/m² · d

Wetland Profiles

Example profiles in dissolved oxygen are shown in Figure 5.3 for a low DO influent to a FWS system in a warm climate (Tres Rios, Arizona). There are not large increases in DO (due to reaeration), nor large decreases (due to WOD). A similar situation prevails for HSSF wetlands, as illustrated in Figure 5.4 (NERCC, Minnesota). These profiles do not resemble the "oxygen sag" profiles of streams subjected to point sources of oxygen demand.

The net oxygen supply rate can be positive (supply), negative (consumption), or zero. The data of Stengel *et al.* (1987) provide values of net oxygen consumption rates for *Phragmites* gravel bed wetlands. Fully oxygenated tap water with zero BOD and zero TKN was fed to the wetland, and the DO was found to decrease with distance in the inlet region. The SSF wetland was thus consuming oxygen in the absence of incoming BOD or NOD, with strong seasonal variations (Figure 5.5).

The interpretation of the data presented in Figure 5.5 is simply that WOD exceeded the transfer of oxygen from air; and DO was depleted. Photosynthetic production of O_2 was likely zero in the gravel bed, and no mass transfer would be expected at the inlet, because the water was saturated with DO. Consequently, the rates shown in Figure 5.5 correspond to $r_{O,\,WOD}$ (see Equation 5.11).

Stengel (1993) also found that after the initial drop in DO, reaeration did not occur; rather, DO reached a stable (constant) value with increasing distance along the bed. Cattails provided a stable root zone DO of about 1–2 mg/L in summer, whereas *Phragmites* stabilized at essentially zero DO. The implication is that in the downstream portions of the wetland, all oxygen uptake was consumed by respiration and SOD. It is important to note that this zero-loaded HSSF wetland was not able to sustain a high oxygen concentration in the water: the internal wetland processes consumed all transferred oxygen.

The stoichiometric coefficients in Equation 5.10 are often taken to be $a_B = 1.5$ and $a_N = 4.5$. However, wetland data sets are not consistent with that presumption (Kadlec and Knight, 1996). When Equation 5.4 was regressed for wetlands with DO, BOD, and NH_4-N information, the stoichiometric coefficients were very much smaller. The inference is that biomass compartments participate in dictating the oxygen level.

It is concluded that the Streeter–Phelps analysis is not suitable for wetlands, due to lack of the ability to quantify wetland oxygen demand (WOD), which is a more dominant factor in wetlands than in streams. It is therefore instructive to summarize some operational results instead. Table 5.1 lists several annual average inlet and outlet DO values for treatment wetlands, together with the associated BOD and ammonia concentrations. It is clear from these examples that HSSF

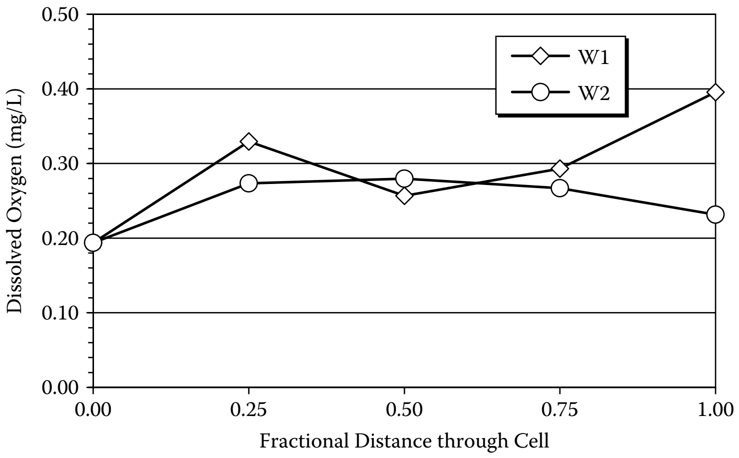


FIGURE 5.4 Dissolved oxygen profiles for the NERCC, Minnesota, HSSF wetlands (W1 and W2). There is essentially no DO in the incoming water, and none along the flow direction including the outlet. There are 31 measurement occasions over two years.

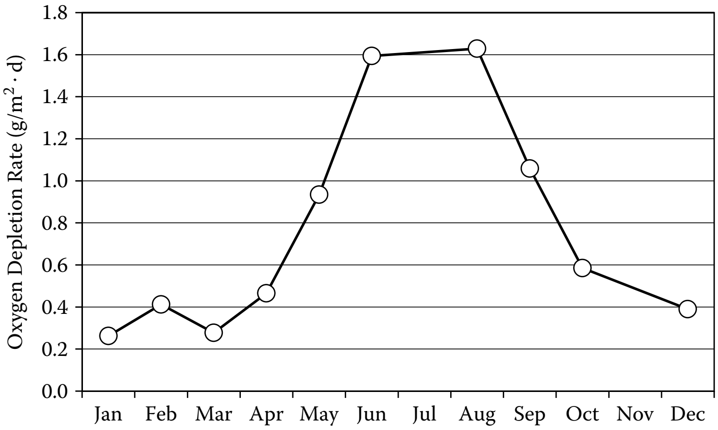


FIGURE 5.5 Oxygen depletion rate in the inlet zone of a *Phragmites* gravel bed wetland receiving oxygenated tap water with nitrate at 30 ± 2 mg/L. (Data from Stengel *et al*. (1987) In *Aquatic Plants for Water Treatment and Resource Recovery*. Reddy and Smith (Eds.), Magnolia Publishing, Orlando, Florida, pp. 543–550.)

TABLE 5.1
Dissolved Oxygen Entering and Leaving Treatment Wetlands

Wetland System	HLR (cm/d)	Inlet BOD (mg/L)	Outlet BOD (mg/L)	Inlet NH_3-N (mg/L)	Outlet NH_3-N (mg/L)	Inlet DO (mg/L)	Outlet DO (mg/L)
Free Water Surface							
Hillsdale, Michigan	0.8	≈ 0	≈ 0	0.01	0.04	9.13	8.82
Commerce Twp., Michigan	18.2	1.18	2.32	0.064	0.050	8.32	9.86
Orlando Easterly, Florida	4.9	1.95	1.02	0.33	0.09	6.10	2.62
Tres Rios, Arizona	10.9	2.26	1.53	1.69	0.75	6.10	2.62
Listowel 3, Ontario	1.3	19.4	7.3	7.04	3.43	5.65	3.48
Augusta, Georgia	7.3	10.47	4.71	2.51	2.15	4.83	7.21
Sacramento, California	6.5	23.9	6.5	15.4	10.4	3.28	2.99
Listowel 4, Ontario	1.8	55.7	9.5	8.80	6.98	2.13	2.71
Richmond, New South Wales Open Water	6.4	51.7	22.9	35.2	17.5	1.01	8.50
Pontotoc 2, Mississippi	1.54	46.5	26.5	112	39	3.57	5.94
Portland, New Zealand	5.2	33	10	1.7	4.9	11.2	5.3
Oregon State 2	3.95	1003	291	168	88	2.39	0.09
Subsurface Flow							
Benton, Kentucky #3	7.1	25.6	6.2	4.8	8.6	8.20	1.00
Richmond, New South Wales Bulrush	5.1	51.7	5.8	35.2	19.5	1.01	0.00
Richmond, New South Wales Cattail	4.6	51.7	4.7	35.2	18.8	1.01	0.04
Richmond, New South Wales Gravel	3.8	51.7	4.3	35.2	19.2	1.01	0.25
Hardin, Kentucky #2	4.9	32.1	4.6	3.4	3.2	3.04	0.60
Minoa, New York	14	149	44	23.2	20.6	4.21	0.03
Grand Lake, Minnesota	1.02	184	69	51.2	24.5	0.10	0.30
NERCC, Minnesota	1.36	256	36	73.5	50.2	0.17	0.33
Břehov, Czech Republic	2.6	109	27	40	24.7	1.4	4.0
Ondřejov, Czech Republic	7.5	104	12	18.3	25.5	5.5	4.9
Čistá, Czech Republic	17.4	37	7.3	14.1	12.8	4.9	3.7
Dušníky, Czech Republic	1.8	716	56	54	27	0.9	2.0
Mořina, Czech Republic	2.8	116	27	35.4	32.3	1.5	0.2
Rector, Arkansas	7.6	45	27	0.7	4.4	5.7	0.7
Smackover, Arkansas	19.4	19	16	3.5	2.2	4.1	0.3
Waldo, Arkansas	20.2	28	14	2.0	3.5	10.2	0.2
Waipoua HQ, New Zealand	0.4	63	11	47.3	35.7	1.1	2.9

Note: Oxygen consumption is to some extent related to the differences between inlet and outlet BOD and ammonia. Subsurface systems are more heavily loaded with BOD and NOD, and have essentially no DO in their effluents.

wetlands in North America commonly do not have any substantial amount of DO in their effluents. Additionally, the intensive studies at the Tennessee Tech site, with 14 HSSF wetlands, in Baxter, Tennessee, found DO essentially at or below the detection limit over a two-year period (George *et al*., 1998).

However, Vymazal and Kröpfelová (2006) reported substantial concentrations of DO at the outflow of many Czech HSSF systems. Out of 59 HSSF wetlands surveyed, they found 33 with outflow DO less than 3 mg/L, and 18 with DO greater than 5 mg/L. The HSSF wetlands receiving dairy wastewater in New Zealand, with high CBOD and ammonia in the inlet, produced moderate DO, in the range of 3–5 mg/L (Tanner *et al*., 1995a; Tanner *et al*., 1998b). According to the oxygen mass balance (Equation 5.12), there should be no DO in HSSF wetland discharges when treating wastewaters with high oxygen demands. Vymazal and Kröpfelová (2006) suggested that outflow DO concentration is a very poor indicator of processes occurring in the SSF wetlands, but the reverse appears to be important as well: Reduction of CBOD and ammonia are not good indicators of the outlet DO.

There are a number of potential reasons for unexpectedly high DO in some HSSF effluents. Reaeration in outlet structures may occur due to splash and exposure to air. The membrane electrode measurement is often used, and is subject to interferences from hydrogen sulfide and from dissolved salts. Preferential flow paths in the wetland, including the possibility of overland flow, can lead to effluents that are not representative of the water within the gravel matrix.

The situation for FWS wetlands is also not clear. Some lightly loaded systems have a great deal of DO (Commerce Township, Michigan), while others do not (Orlando, Florida Easterly; Tres Rios, Arizona). Some with moderate loading reaerate to a large extent (Richmond, New South Wales Open Water; Pontotoc, Mississippi).

It is of interest to compare the open water and gravel systems at Richmond, New South Wales. These had the same geometry, received the same influent water, and both were devoid of macrophytes. BOD and ammonia were reduced in both (Table 5.1). The open water system had fully aerated water at the outlet, whereas the gravel bed effluent was very low in DO. The conclusion may be drawn that the presence of gravel interfered with oxygen transfer.

The Sediment–Water Interface

Dissolved oxygen uptake at a sediment–water interface (SOD) is controlled by mass transport and/or biochemical reactions in two adjacent boundary layers: the diffusive boundary layer in the water and the penetration in the sediment (Higashino *et al*., 2004). Those boundary layers are very thin, with dimensions measured in millimeters (Crumpton and Phipps, 1992). As a result of the slow rate of oxygen transport through interstitial water and a comparatively high oxygen demand, the surface oxidized soil or sediment horizon is thin and ranges from a few millimeters to a few centimeters in depth, depending on the oxygen consumption capacity of the material. Though this oxidized surface horizon is thin, biological and chemical processes occurring in this zone strongly influence the availability of both nutrients and toxins in flooded soils and sediment–water interface (Gambrell and Patrick, 1978).

Under FWS wetland conditions, there is a strong dependence of SOD exertion on velocity, and transport through the diffusive boundary layer is limiting.

Vertical Stratification

Vertical dissolved oxygen profiles have not been extensively studied in treatment wetlands. However, results from three types of systems help provide insights: ponds, wetlands with submerged aquatic vegetation (SAV), and HSSF wetlands. All three of these variants of treatment wetlands exhibit vertical stratification with respect to oxygen.

Pond studies have shown some variable but strong vertical gradients over the top 25 cm of the water column (Abis, 2002). Because concentrations often exceed saturation in the top pond water layer, algal photosynthetic reaeration is present. The high values of DO at the water surface are caused by the preferential interception of photosynthetically active radiation (PAR) in the upper water layers.

Given that physical transfer occurs from the atmosphere, and biochemical generation can occur within the water column, vertical profiles of DO are anticipated in FWS wetlands, and in fact are found in the field. Extensive measurements were made in the lightly loaded treatment wetlands of the Everglades, Florida, Nutrient Removal Project (Chimney *et al*., 2006) (Figure 5.6). The highest DO values were found in the open water and submerged vegetation zones, with a strong decreasing gradient with depth. In contrast, DO values in areas of floating plants and emergent vegetation were low, only 1–2 mg/L on average. FWS wetlands with submerged aquatic vegetation display strong vertical profiles of DO (Table 5.2). This is presumably also due to photosynthetic reaeration, with the submerged macrophytes proving oxygen, rather than algae. As in algal ponds, the upper water zones are preferentially active.

Vertical profiles of DO in HSSF wetlands are also present, but with much lesser values and smaller gradients (Table 5.2). HSSF wetlands typically have very little or no dissolved oxygen anywhere in the water column (Table 5.2). Neither algae nor SAV are present to contribute to photosynthetic reaeration. Physical reaeration can and does occur, but transfer rates are lessened by the presence of the gravel media, which precludes wind enhancement and lengthens diffusion distances. As a consequence, oxidation-reduction potential (E_h) (see the following section of this chapter) becomes a more effective measure of conditions within the bed. Nominally, negative E_h values correspond to the absence of DO, and provide conditions conducive to reduction of nitrate, iron, and sulfate (Reddy and D'Angelo, 1994). For HSSF wetlands, physical reaeration from the top represents the dominant mechanism. Comparison of planted and unplanted beds shows that there is essentially no effect of vegetation, with the vegetated systems at Minoa, New York, and Vilagrassa, Spain, showing slightly lower E_h than the unvegetated systems.

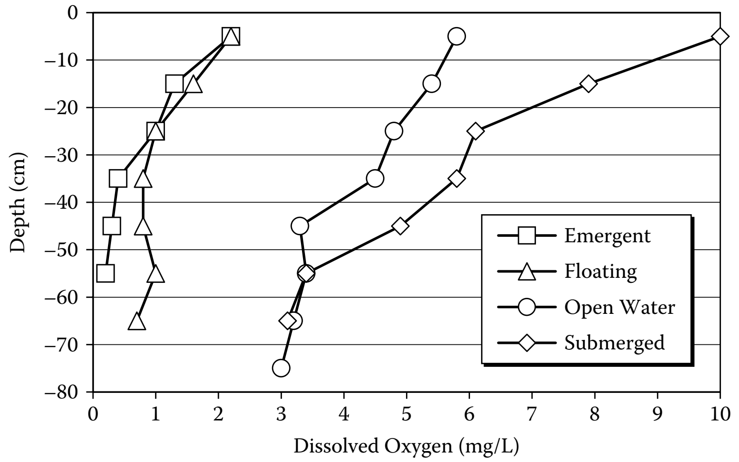


FIGURE 5.6 Vertical profiles of dissolved oxygen in the various vegetation types in the Everglades Nutrient Removal Project FWS wetlands, Florida. Data are from 141 profiles collected over a 2.5-year period. (Data from Chimney *et al.* (2006) *Ecological Engineering* 27(4): 322–330.)

TABLE 5.2
Vertical E_h and DO Profiles in Treatment Wetlands

HSSF System		Bed Depth (cm)	Bottom (cm)	Mid (cm)	Mid (cm)	Top (cm)
Grand Lake		60	53	—	23	8
	DO	mg/L	0.24	—	17.9	0.49
NERCC 1		45	40	—	23	—
	DO	mg/L	0.11	—	0.16	—
NERCC 2		45	40	—	23	—
	DO	mg/L	0.08	—	0.13	—
Minoa Planted		84	70	—	40	10
	DO	mg/L	0.02	—	0.06	0.47
	E_h	mv	−243	—	−229	−192
Minoa Unplanted		84	70	—	40	10
	DO	mg/L	0.04	—	0.03	0.20
	E_h	mv	−238	—	−218	−194
Vilagrassa Planted		70	30	20	10	0
	Inlet E_h	mv	−115	−120	−70	—
	Outlet E_h	mv	−25	−10	70	160
Vilagrassa Unplanted		70	30	20	10	0
	Inlet E_h	mv	−90	−80	−55	60
	Outlet E_h	mv	−5	5	100	160
FWS SAV System		**Water Depth (cm)**	**Bottom (cm)**	**Mid (cm)**	**Mid (cm)**	**Top (cm)**
Arcata		100	90	—	50	10
	DO	mg/L	0.5	—	6	11
ENR Shallow		40	—	—	30	3
	DO	mg/L	—	—	7.0	12.3
ENR Medium		80	—	60	30	3
	DO	mg/L	—	3.9	4.2	13.8
ENR Deep		120	90	60	30	3
	DO	mg/L	7.3	7.5	9.5	15.2

Source: For data on HSSF: for Grand Lake and NERCC, Minnesota: unpublished data; for Minoa, New York: Theis and Young (2000) *Subsurface flow wetland for wastewater treatment at Minoa*. Final Report to the New York State Energy Research and Development Authority, Albany, New York; for Vilagrassa, Spain: García *et al.* (2003a) *Ecological Engineering* 21(2–3): 129–142. For data on FWS: for Arcata, California: U.S. EPA (1999) *Free water surface wetlands for wastewater treatment: A technology assessment*. EPA 832/R-99/002, U.S. EPA Office of Water: Washington, D.C. 165 pp.; for ENR, Florida mesocosms: DBE (1999) *A demonstration of submerged aquatic vegetation/limerock treatment system technology for removal of phosphorus from Everglades agricultural area water: Final Report*. Prepared for the South Florida Water Management District (SFWMD) and the Florida Department of Environmental Protection (FDEP). Contract No. C-E10660, DB Environmental (DBE).

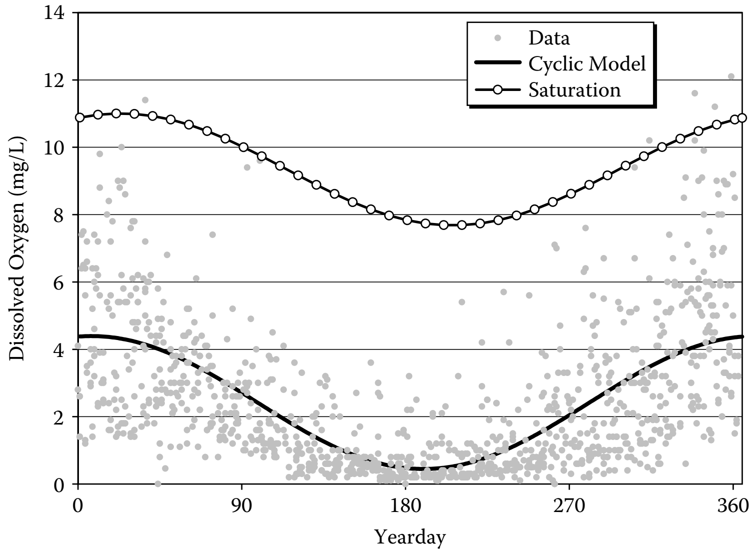


FIGURE 5.7 Annual progression of dissolved oxygen at the Tres Rios, Arizona, FWS Hayfield wetlands. Six years' data are represented for two wetlands (H1 and H2), at an average detention time of 5.3 days.

Trends and Variability

The annual temperature cycle in FWS systems creates a similar cycle in the saturation concentrations of dissolved oxygen, with greater solubility in the colder months. Consequently, the driving force for physical reaeration is maximum in cold months. The photosynthetic production of oxygen in the water column, by algae and/or submerged macrophytes, is driven by a seasonal cycle in solar radiation (PAR). It is therefore expected that wetland water dissolved oxygen, if any, will follow a seasonal cycle with larger values in cold months. This is indeed the case for those systems that have been studied, such as the Tres Rios, Arizona, wetlands (Figure 5.7). Equation 6.1 (see Chapter 6 for a full discussion of this equation) was fit to the DO data. The annual trend in daily values at Tres Rios had an amplitude of about 80% of the annual mean of 2.4 mg/L, with the maximum in January. Cyclic trends are similar in other FWS wetlands, with the parameters depending on location and loading (Table 5.3).

$$C = C_{avg}\left[1 + A\cdot\cos\left(\omega(t - t_{max})\right)\right] + E \qquad (6.1)$$

The values of E in Equation 6.1 follow a distribution that is nearly normal (Figure 5.8). The breadth of the scatter changes during the course of the year, with more scatter in the winter. The median amplitude of the annual cycle is 65% of the annual mean for FWS wetland outflows (Table 5.3).

TABLE 5.3
Trend Multipliers for Dissolved Oxygen in FWS Wetlands

Wetland System	Years	Mean	Amplitude	Yearday Maximum	R^2	Excursion Frequency 5%	10%	20%	50%
Orlando, Florida Easterly Wetland	10	2.64	0.41	21	0.213	0.44	0.53	0.74	1.04
Hillsdale, Michigan EA	6	6.92	0.65	32	0.370	0.08	0.17	0.35	1.00
Hillsdale, Michigan ET	6	6.94	0.65	32	0.368	0.08	0.17	0.35	1.00
Hillsdale, Michigan WT	6	8.65	0.78	43	0.509	1.08	1.17	1.35	2.00
Hillsdale, Michigan WA	6	8.70	0.61	44	0.392	0.05	0.11	0.59	0.97
Tres Rios, Arizona, Hayfield 1	6	2.54	0.91	10	0.353	0.21	0.29	0.40	0.79
Tres Rios, Arizona, Hayfield 2	6	2.29	0.72	4	0.356	0.28	0.35	0.46	0.81
Tres Rios, Arizona, Cobble 1	6	3.29	0.84	364	0.280	0.17	0.20	0.31	0.65
Tres Rios, Arizona, Cobble 2	6	2.77	0.79	11	0.278	0.16	0.23	0.31	0.65
Listowel, Ontario, 3	4	3.51	0.63	360	0.285	0.06	0.13	0.26	0.89
Musselwhite, Ontario	4	5.33	0.65	42	0.351	0.50	0.59	0.71	0.93
Titusville, Florida	7	2.55	0.38	33	0.439	0.55	0.68	0.78	1.02
ENRP, Florida	6	3.7	0.44	41	0.418	0.34	0.52	0.72	0.99
Commerce Township, Michigan	4	10.23	0.20	61	0.602	0.83	0.87	0.92	0.98
Median			**0.65**			**0.25**	**0.32**	**0.52**	**0.98**

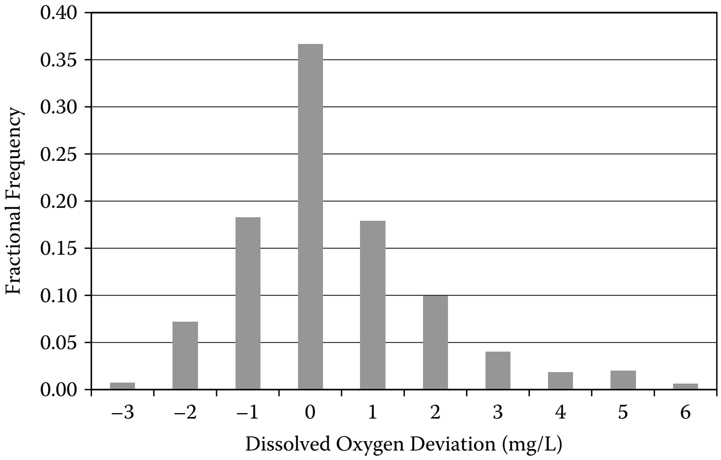


FIGURE 5.8 Variation about the mean trend for dissolved oxygen leaving the Tres Rios, Arizona, FWS Hayfield wetlands (H1 and H2). See Figure 5.7 for the annual times series.

The median time of the maximum in outflow DO is early February (yearday = 32, Table 5.3).

Vymazal and Kröpfelová (2006) found little seasonal variation in the DO in the outflow of a number of Czech HSSF wetlands. The same is true of the various HSSF wetlands in the United States that do not display any measurable DO in the outflow.

The percentile points of the DO scatter around the annual cosine trends are given in Table 5.3. It is seen that with some frequency, the excursions from the trend DO values are lower by a considerable margin. For instance, 5% of the time, the median DO is only 25% of the trend value (Table 5.3). This means that none of the example FWS systems in Table 5.3 satisfy the United States DO requirement for discharge to receiving waters at the 95% level of confidence (greater than 5 mg/L 19 times out of 20). This means that extra design features (such as cascade aeration) must be implemented to meet the DO requirement for surface discharges. The same conclusion would be reached for HSSF wetlands, certainly in the United States, but also in the broader context of all HSSF wetlands.

5.3 VOLATILIZATION

Although oxygen transfer is a critical feature of treatment wetlands, there are several other gases that transfer to and from the ecosystem. Incoming volatile anthropogenic chemicals may be lost. But a treatment wetland also takes in atmospheric carbon dioxide for photosynthesis, and expels it from respiratory processes. The various treatment processes create product gases, which are also expelled from the wetland. These include ammonia, hydrogen sulfide, dinitrogen, nitrous oxide, and methane. Of these, carbon dioxide, nitrous oxide, and methane are regarded as greenhouse gases, and are of concern as atmospheric pollutants. As a result, there have been several treatment wetland studies focused on these three gases. Volatilization of ammonia is discussed in Chapter 9, and volatilization of hydrogen sulfide in Chapter 11.

Methane is produced by anaerobic processes with the wetland substrate. Carbon dioxide is produced by aerobic microbial processes, and by root respiration. Nitrous oxide is a possible product of (incomplete) denitrification. Because these greenhouse gases contribute to global warming, they have received attention in the context of treatment wetlands (Brix *et al.*, 2001; Teiter and Mander, 2005).

Nitrous Oxide

Denitrification typically proceeds through a sequence of steps, ultimately leading to formation of dinitrogen (see Chapter 9). An intermediate product is N_2O, which may be emitted prior to complete reduction. Partial oxidation of ammonia (partial nitrification) is another candidate mechanism for N_2O formation. ^{15}N experiments have sometimes shown that this reaction is not dominant (Itokawa *et al.*, 2001), but in other circumstances have identified partial nitrification as the primary source (Beline *et al.*, 2001).

N_2O is stable in the atmosphere, with a lifetime of over 100 years. It also is a major contributor to global warming, with a carbon dioxide equivalency of about 300. A number of studies have used chamber assay methods to measure N_2O emission in treatment wetlands, both FWS (Freeman *et al.*, 1997; Gui *et al.*, 2000; Johansson *et al.*, 2003; Mander *et al.*, 2003; Johansson *et al.*, 2004; Hernandez and Mitsch, 2005; Søvik *et al.*, 2006; Liikanen *et al.*, 2006); and SSF (Kløve *et al.*, 2002; Mander *et al.*, 2003; Teiter and Mander, 2005; Søvik *et al.*, 2006). The rates of emission average about 4,000 $\mu gN/m^2 \cdot d$ for 15 wetlands, which amounts to an average of 2.2% of the nitrogen load removed in the wetlands (Table 5.4).

Denitrification is strongly seasonal, with larger rates in warm seasons, therefore it is not surprising that nitrous oxide emission is also seasonal, with maxima in summer (Teiter and Mander, 2005; Hernandez and Mitsch, 2005). However, Johansson *et al.* (2003) found no seasonality at the Nykvarn FWS treatment wetlands near Linköping, Sweden.

TABLE 5.4
Greenhouse Gas Emissions from Treatment Wetlands

Wetland Type and Country	Location	Details	Reference	CO_2-C Emission Rate ($gC/m^2 \cdot d$)	CH_4-C Emission Rate ($mgC/m^2 \cdot d$)	Estimated % of Load Removed	N_2O-N Emission Rate ($\mu gN/m^2 \cdot d$)	Estimated % of Load Removed
FWS								
China	Liaohe Delta	Summer, natural	Huang *et al.* (2005)	—	—	—	41,000	—
Estonia	Kodijärve	Summer	Søvik *et al.* (2006); Mander *et al.* (2003)	0.96	340	—	7,100	0.66
Finland	Hovi	Summer	Søvik *et al.* (2006)	0.21	29	−250	400	7.6
Finland	Kompsasuo	Infilt	Søvik *et al.* (2006); Liikanen *et al.* (2006)	0.73	310	390	190	0.29
Finland	Lakeus	Summer	Søvik *et al.* (2006)	2.00	350	57	350	0.17
Finland	Ruka	Infiltration	Søvik *et al.* (2006)	1.30	72	59	4,900	0.95
Norway	Skjønhaug	Summer	Søvik *et al.* (2006)	0.98	—	13	4,000	0.28
China	Jiaonan	—	Gui *et al.* (2000)	—	19	0.13	4,000	0.10
Sweden	Ormastorp	SAV	Stadtmark and Leonardson (2005)	—	240	—	—	—
Sweden	Görarp	SAV	Stadtmark and Leonardson (2005)	—	240	—	—	—
Sweden	Genarp	SAV	Stadtmark and Leonardson (2005)	—	240	—	—	—
Sweden	Nykvarn	—	Heiberg (1999); Johansson *et al.* (2003; 2004)	—	135	—	1,985	0.37
Wales	Cerig-yr-Wyn	—	Freeman *et al.* (1997)	—	—	—	233	—
United States	Columbus, Ohio	—	Hernandez and Mitsch (2005)	—	—	—	92	0.48
HSSF								
Estonia	Kõo	Summer	Sovik *et al.* (2006); Mander et al. (2003)	0.38	160	—	4,200	0.17
Norway	Ski	Summer	Sovik *et al.* (2006)	0.26	130	24	6,900	3.3
Poland	Nowa Slupia	Summer	Sovik *et al.* (2006)	0.56	670	—	—	—
Denmark	Kalø	—	Brix (1990)	0.56	220	6.8	—	—
New Zealand	Hamilton	High, Veg, Up	Tanner *et al.* (1997)	—	142	9.0	—	—
New Zealand	Hamilton	High, Veg, Down	Tanner *et al.* (1997)	—	34	2.1	—	—
New Zealand	Hamilton	Low, Veg, Up	Tanner *et al.* (1997)	—	116	12.3	—	—
New Zealand	Hamilton	Low, Veg, Down	Tanner *et al.* (1997)	—	65	6.9	—	—
New Zealand	Hamilton	M1	Tanner *et al.* (2002)	2.62	378	0.36	—	—
New Zealand	Hamilton	D1	Tanner *et al.* (2002)	1.98	141	1.67	—	—
New Zealand	Hamilton	D2	Tanner *et al.* (2002)	1.09	103	3.24	—	—
New Zealand	Hamilton	D2A	Tanner *et al.* (2002)	1.55	103	12.13	—	—
Norway	Jordforsk	Experiment 6	Kløve *et al.* (2002)	—	—	—	890	0.06
VF								
Estonia	Kõo	Summer	Sovik *et al.* (2006); Mander *et al.* (2003)	1.60	110	—	15,000	0.28
Norway	Ski	Summer	Sovik *et al.* (2006)	3.90	140	0.63	9,600	16
Mean				**1.29**	**187**	**20.5**	**3,989**	**2.2**
Median				**1.03**	**141**	**6.9**	**4,000**	**0.3**

TABLE 5.5
Gas Emissions in Different Plant Communities in the Nykvarn, Sweden, FWS Treatment Wetland

Plant Community	N	CH_4 Flux (mg/m²·d)	N_2O Flux (mg/m²·d)
Typha latifolia	146	163	3.84
Phalaris arundinacea	12	318	5.95
Spirogyra spp.	111	168	1.53
Glyceria maxima	37	160	1.19
Lemna minor	4	675	2.27
No plants	15	245	5.95

Source: Data from Johansson *et al.* (2003) *Tellus* 55B: 737–750; Johansson *et al.* (2004) *Water Research* 38: 3960–3970.

There is also potentially an effect of the particular plant community on N_2O emissions (Table 5.5). At the Nykvarn, Sweden, site, studies showed that plants generally reduced N_2O emissions, but the opposite was found at the Olentangy site in Columbus, Ohio.

Methane

Methanogenesis occurs frequently in the sediment layers of treatment wetlands, particularly HSSF systems, and particularly in wetlands receiving high loads of CBOD. Carbohydrates from various sources are broken down by fermentation, forming low molecular weight compounds which are then further broken down into methane and water by methanogenic bacteria (Equation 5.21). The methane so formed may either be oxidized, or exit the wetland via plant airways or volatilization from sediments and water (Figures 5.9 and 5.10).

Methane is stable in the atmosphere, with a lifetime of over eight years. It also is a major contributor to global warming, with a carbon dioxide equivalency of about 23. A number of studies have used chamber assay methods to measure CH_4 emission in treatment wetlands, both FWS (Gui *et al.*, 2000; Johansson *et al.*, 2003; Mander *et al.*, 2003; Johansson *et al.*, 2004; Søvik *et al.*, 2006; Liikanen *et al.*, 2006); and SSF (Brix, 1990; Tanner *et al.*, 1997; Kløve *et al.*, 2002; Tanner *et al.*, 2002a; Mander *et al.*, 2003; Teiter and Mander, 2005; Søvik *et al.*, 2006). The rates of emission average about 187 mgC/m²·d for 24 wetlands, which amounts to an average of 20% of the carbon load removed in the wetlands (Table 5.4).

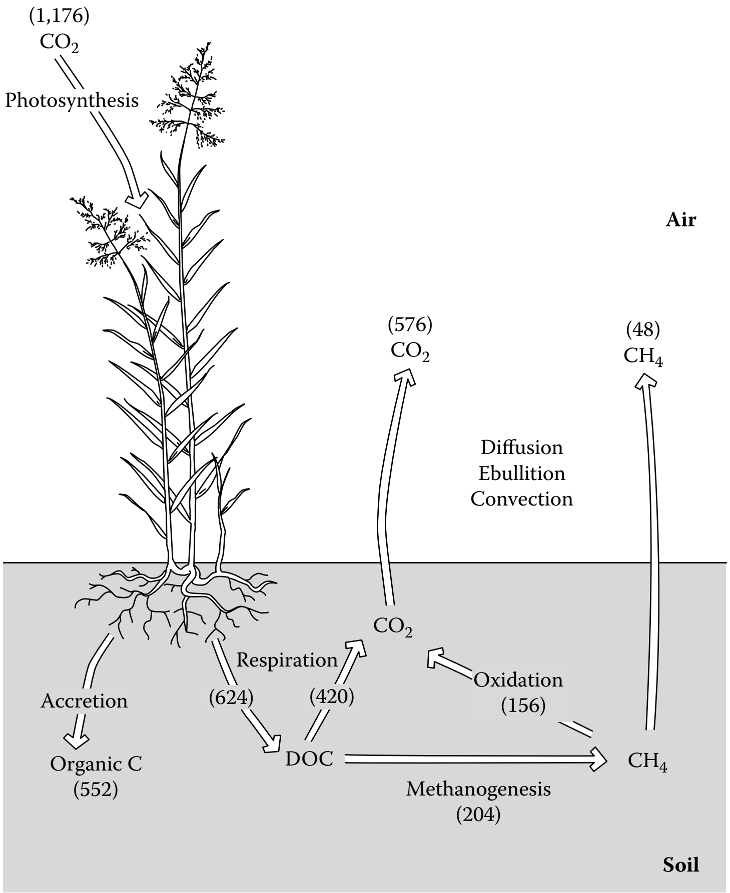


FIGURE 5.9 Carbon processing and gas emission in treatment wetlands. The numbers are fluxes in gC/m²·yr, as measured for a *Phragmites* stand at the Vejlerne Nature Preserve in Denmark. Inflows and outflows of carbon with water are minimal in this natural wetland. By comparison with values in Table 5.4, these numbers are not far different from treatment wetland values. (Redrawn from Brix *et al.* (2001) *Aquatic Botany* 69: 313–324. Reprinted with permission.)

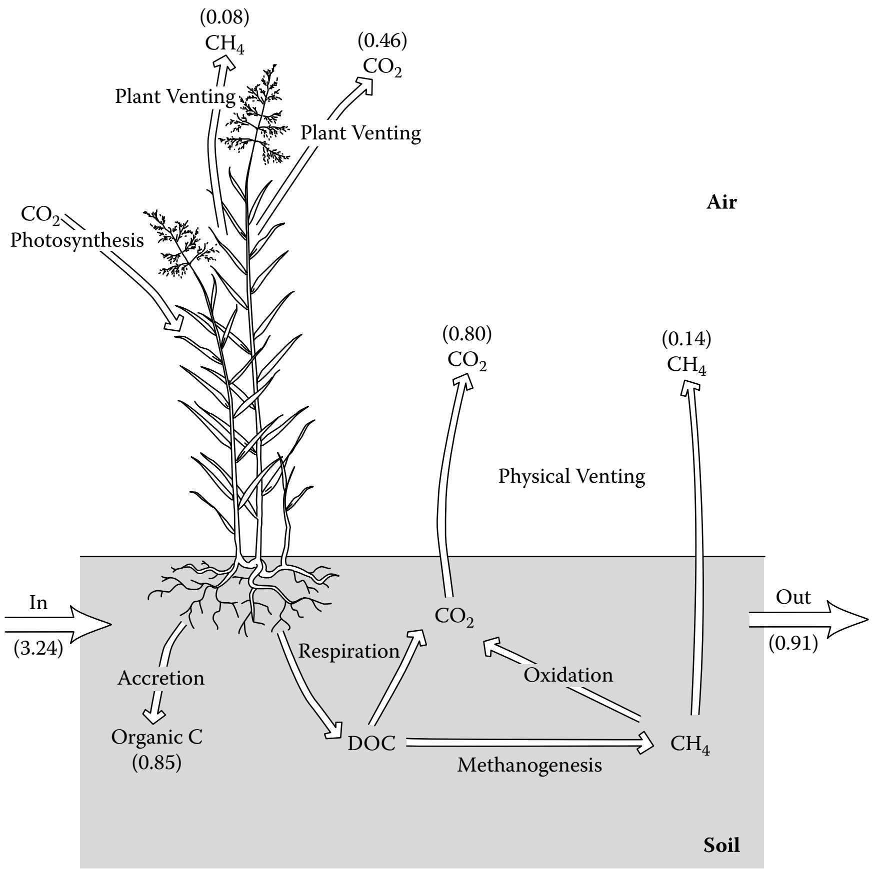


FIGURE 5.10 Carbon processing and gas emission in the HSSF treatment wetlands at Kalø, Denmark, in April. The numbers in italics are fluxes in $gC/m^2 \cdot d$. The *Phragmites* stand was in a senesced state. (Redrawn from Brix (1990) *Water Research* 24(2): 259–266. Reprinted with permission.)

There is also potentially an effect of the particular plant community on CH_4 emissions (Table 5.5). At the Nykvarn, Sweden, site, studies showed that some plants reduced CH_4 emissions, but others showed greater emission, compared to zones with no plants. Sorrell and Boon (1992) found that plants slightly reduced the methane emissions measured in a natural Australian wetland.

Part of the reason for differences from one plant community to another has to do with the various mechanisms of gas exchange (see Figure 5.10). The airways associated with emergent plants function as both influx and efflux conduits for gases (Sorrell and Armstrong, 1994). Live plant culms can serve either function, and indeed adjacent culms attached to the same rhizome may serve opposite functions. Standing dead (and perhaps broken) stems or culms can also transport gases. Figure 5.10 shows the losses from a *Phragmites* HSSF wetland in Denmark during April, when standing dead culms dominated the reedbed. A substantial proportion of methane loss was via the plants. The zones between plants provide for the loss of gases by volatilization from water and substrate. Obviously, the plant density affects the relative proportions of the two mechanisms.

Methane emission is strongly seasonal, with larger rates in warm seasons (Figure 5.11) (Johansson *et al.*, 2004; Teiter and Mander, 2005). Reports from treatment wetland studies are reinforced by results from studies on natural wetlands, such as those of Sorrell and Boon (1992).

Wetlands exhibit strong longitudinal gradients in carbon compounds, as treatment proceeds in the flow direction. It is therefore expected that there should be gradients in methane generation, and indeed that is the case (Figure 5.12).

Carbon Dioxide

Carbon dioxide is utilized by plants and algae in photosynthesis. It is produced by respiration in the root system of plants, and by microbial processes in soils and sediments. Oxidation of carbonaceous components of waters is largely dissipated by oxidation to CO_2. As a result, large fluxes of CO_2 are present in wetlands, some as influxes to the green plants, and some as releases. Figure 5.9 illustrates an approximate annual mass balance for carbon in a *Phragmites* wetland that is not receiving any wastewater. Approximately 50% of the net annual photosynthesis CO_2 fixation is ultimately respired to CO_2 and CH_4 in the sediment, but only small proportions are directly released to the atmosphere (Brix *et al.*, 2001). The fixation of atmospheric carbon dioxide is synchronous with the growing season. The moderately large standing crops of biomass require on the order of 1,000–2,000 $gC/m^2 \cdot yr$.

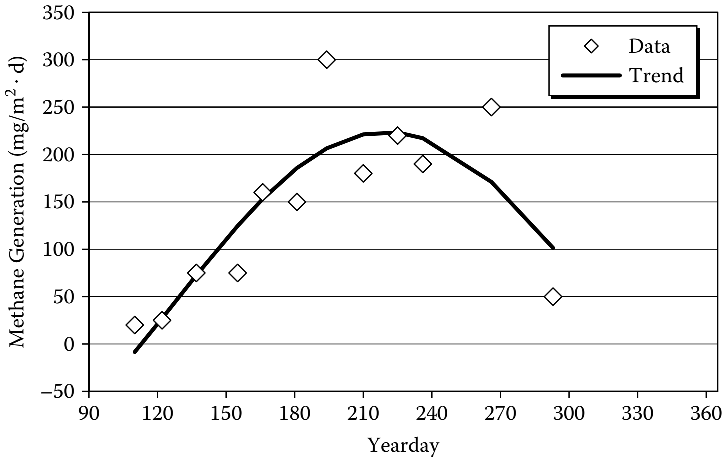


FIGURE 5.11 Seasonal trend in methane production from the Nykvarn, Sweden, FWS treatment wetland. (From Johansson *et al.* (2004) *Water Research* 38: 3960–3970. Reprinted with permission.)

Untreated municipal wastewaters have ratios of TOC to CBOD of 0.5–0.8, settled wastewaters are 0.8–1.2, and treated effluents are 2–5 (Metcalf and Eddy, Inc., 1991; Crites and Tchobanoglous, 1998). Treatment wetlands receiving secondary, tertiary, and lagoon waters have ratios of TOC to CBOD of 5–10 (see Table 8.1). Loadings of BOD are typically in the range of 40–4,000 g/m²·yr, and thus carbon loadings are roughly 100–10,000 gC/m²·yr. Consequently, either atmospheric fixation or influent carbon loadings may be dominant in a treatment wetland. FWS wetlands treating secondary or tertiary effluents would fixation-dominated, whereas systems treating septic tank effluents would be influent-dominated with respect to carbon.

As for nitrous oxide and methane, part of the emitted CO_2 is lost through plant airways, and part via losses from the soil and water air interfaces (Figure 5.10). The rates of emission average about 1.3 gC/m²·d (500 gC/m²·yr) for 16 wetlands (Table 5.4). As noted by Brix (1990), it is difficult to generalize about how much of the incoming carbon load is dissipated to carbon dioxide, because of the interactions of CO_2 and CH_4 in methanogenesis (Equations 5.20 and 5.21), and because of the dual sources of incoming water and the atmosphere. Nonetheless, the amounts of CO_2 emitted to the atmosphere are not trivial compared to those loadings.

Greenhouse Effects

Treatment wetlands sequester organic carbon via the accretion of new sediments and soils. However, they also emit greenhouse gases, CO_2, CH_4, and N_2O. The large multipliers for the radiative effect comparison (300 for nitrous oxide and 20 for methane) mean that small emissions of these gases can counteract the carbon sequestration function. Thus, although wetlands in general, including constructed wetlands, can act as carbon sinks, they still can increase the greenhouse effect because of their release of methane and nitrous oxide (Brix *et al.*, 2001). Because of the small acreage of treatment wetlands compared to natural wetlands, constructed systems are "not so remarkable" as sources of greenhouse

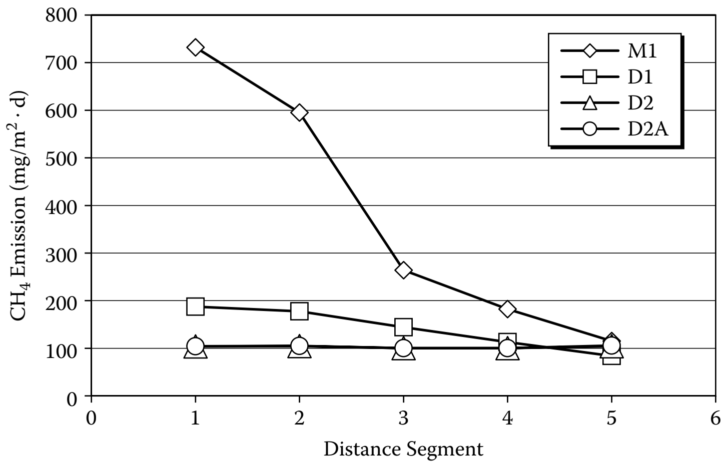


FIGURE 5.12 Methane emissions from four SSF wetlands as a function of distance. Systems M1, D1, D2, and D2A treated different strengths of wastewater. (Data from Tanner *et al.* (2002a) *Ecological Engineering* 18(4): 499–520.)

gases (Mander *et al.*, 2003). Liikanen *et al.* (2006) estimate that even if all global wastewater were treated in constructed wetlands, their share in atmospheric liability would be less than 1% of the total.

5.4 OXIDATION-REDUCTION POTENTIAL

Oxidation-reduction is a chemical reaction in which electrons are transferred from a donor to an acceptor. The electron donor loses electrons and increases its oxidation number or is oxidized; the acceptor gains electrons and decreases its oxidation number or is reduced. The driving force of a chemical reaction is the tendency of the free energy of the system to decrease until, at equilibrium, the sum of the free energies of the products equals that of the remaining reactants. In a reversible oxidation-reduction reaction, this driving force can be measured in Joules or in (milli)volts. Consider a reaction in which *n* electrons are transferred:

$$\mathrm{Ox} + n\,\mathrm{e}^- \rightleftharpoons \mathrm{Red} \tag{5.13}$$

If the free energy change, represented in voltage, is measured against the standard hydrogen electrode, it is denoted by E_h. The equilibrium relation is then:

$$E_h = E_o + \frac{RT}{nF}\ln\left(\frac{[\mathrm{Ox}]}{[\mathrm{Red}]}\right) \tag{5.14}$$

where

E_o = reference potential, mV (zero for the standard hydrogen electrode)
E_h = oxidation reduction potential, mV
F = Faraday's constant, 96.4 J/mol·mV
n = number of electrons transferred
R = gas constant, 8.314 J/mol·°K
T = temperature, °K

and in which the brackets denote concentrations. The interested reader may find more details in chemistry references, such as Ponnamperuma (1972), Pankow (1991), or Morel and Hering (1993).

E_h is a quantitative measure of the tendency of a given system to oxidize or reduce susceptible substances. E_h is positive and high in strongly oxidizing systems; it is negative and low in strongly reducing systems.

Oxidation-reduction conditions affect chemical and microbial processes, and have a very large effect on the biological availability of major and trace nutrients in soils in general (Patrick *et al.*, 1985; Gambrell *et al.*, 1987).

In submerged sediments and soils, redox potential ranges from around −400 mV (strongly reduced) to +700 mV (well oxidized). The oxidation of organic matter yields energy; the amount of energy depends on the nature of oxidant, or electron acceptor. Energetically, the most favorable oxidant is oxygen; after oxygen is depleted there follows a succession of organisms capable of reducing NO_3^-, MnO_2, FeOOH, SO_4^{2-}, and CO_2, with each oxidant yielding successively less energy for the organism mediating the reaction (Westall and Stumm, 1980). This succession leads to zonation, either in the vertical direction with depth into sediments in FWS wetlands, or in the radial direction around roots. The former case is illustrated in Figure 5.13, in which upper layers of the wetland bed display the more energetic reaction zones (Reddy and D'Angelo, 1994). It should be noted that the intermediate zones of Figure 5.13, in which the transition from oxic to anaerobic conditions occurs, are thin in FWS wetlands, typically comprising no more than one or two centimeters. Depending on the magnitude of the vertical transpiration flow, this zone thickness is controlled by downward advection of surface water and its redox potential, with much lesser contributions from diffusion. However, the zonation around wetland plant roots is much smaller still, with typical zone thicknesses of a millimeter or two (Figure 5.14). In HSSF wetlands, the dominant flow is through and under the rhizosphere, and therefore one or more zones may occupy most of the bed thickness (Table 5.2). The direction of supply of oxidants is transverse to the flow direction.

The chemistry of these thin transition layers maybe summarized in a number of equivalent ways (Reddy and D'Angelo, 1994; Mitsch and Gosselink, 2000b); here, a simple version is chosen with organic matter represented by CH_2O. Oxygen is the terminal electron acceptor in aerobic zones, and is reduced while electron donors are being oxidized, notably organic substances and ammonia. This reduction of O_2 to H_2O is carried out by true aerobic microorganisms, and CO_2 is evolved as a waste product:

$$CH_2O + O_2 \rightarrow CO_2 + H_2O \tag{5.15}$$

As O_2 is depleted, nitrate will be used as electron acceptor followed by oxidized manganese compounds and then followed by ferric iron compounds. The order of these reductions is the same as that indicated by thermodynamic considerations (Reddy *et al.*, 1986).

Nitrate is the next oxidant to be reduced following oxygen depletion. Many microorganisms can utilize NO_3^- as terminal hydrogen acceptor instead of O_2, which is the denitrification process (see Chapter 9):

$$5CH_2O + 4NO_3^- \rightarrow 2N_2 + 4HCO_3^- + CO_2 + 3H_2O \tag{5.16}$$

As the redox potential continues to decrease, manganese is transformed from manganic to manganous compounds at about +200 mV (Laanbroek, 1990):

$$CH_2O + 3CO_2 + H_2O + 2MnO_2 \rightarrow 2Mn^{2+} + 4HCO_3^- \tag{5.17}$$

When the reduction of nitrate stops by depletion of this electron acceptor, the reduction of ferric oxide starts. A wide range of anaerobic bacteria are able to conserve energy through the reduction of Fe^{3+} to Fe^{2+} (Laanbroek, 1990; Younger *et al.*, 2002). Many of these microorganisms also have the ability to grow through the reduction of Mn^{4+} to Mn^{2+}.

$$CH_2O + 7CO_2 + 4Fe(OH)_3 \rightarrow 4Fe^{2+} + 8HCO_3^- + 3H_2O \tag{5.18}$$

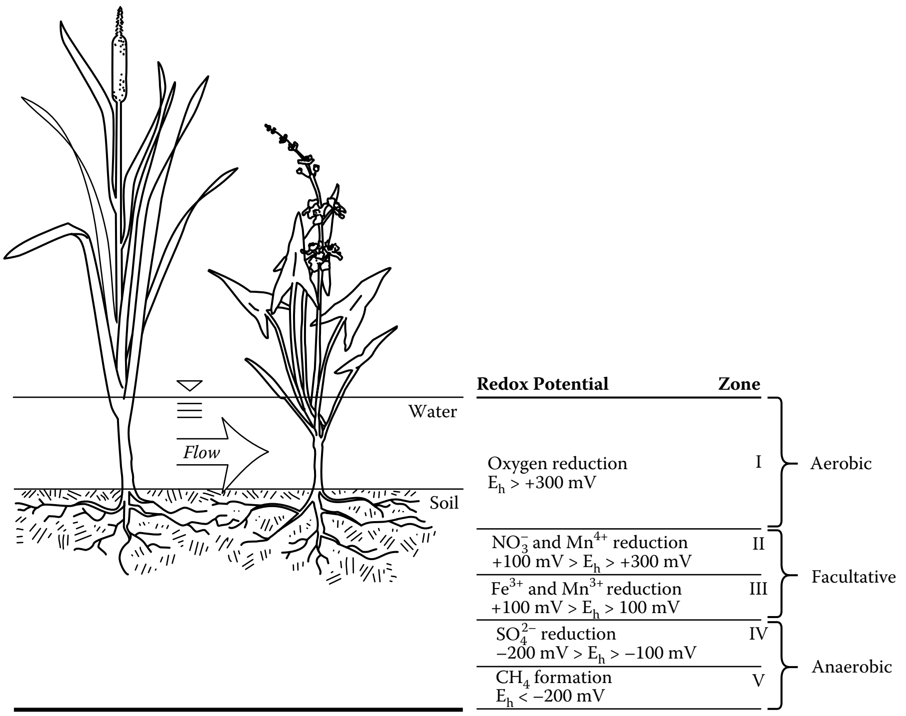


FIGURE 5.13 Hypothetical vertical redox zonation in the soils under a FWS wetland.

Sulfate reduction occurs when the redox potential drops below –100 mV. Only a small amount of reduced sulfur is assimilated by the organisms, and virtually all is released into the external environment as sulfide (Wake *et al.*, 1977).

$$CH_2O + SO_4^{2-} \rightarrow H_2S + 2HCO_3^- \tag{5.19}$$

Sulfate reduction is promoted by design in wetlands built to remove metals with insoluble sulfides (Younger, 2000).

Methane production requires extremely reduced conditions, with a redox potential below –200 mV, after other terminal electron acceptors have been reduced.

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O \tag{5.20}$$

$$4H_2 + CH_3COOH \rightarrow 2CH_4 + 2H_2O \tag{5.21}$$

Methanogenic bacteria utilize hydrogen as an electron source, but can also use formate ($HCOO^-$) or acetate (CH_3COO^-) (Equation 5.21). Methane is either released to the atmosphere or is oxidized to CO_2 by methanotrophic bacteria as soon as it enters the oxic zone.

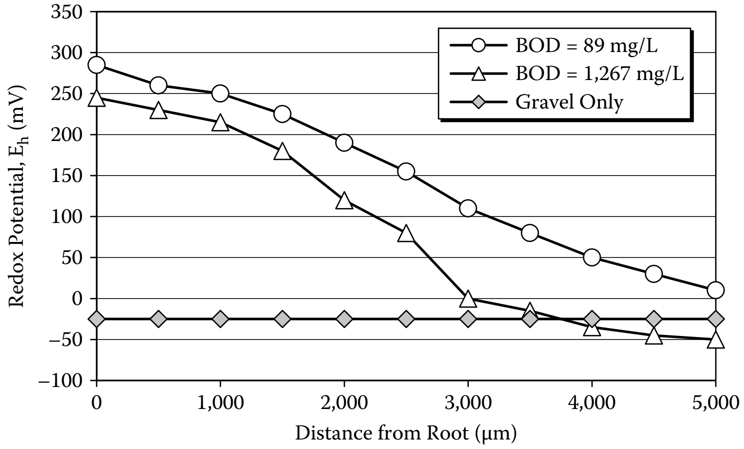


FIGURE 5.14 Profiles of redox in the vicinity of main roots of *Schoenoplectus* (*Scirpus*) *validus* in a HSSF gravel bed wetland, along with an unvegetated control. These profiles were determined via micro-electrodes. Dissolved oxygen at the root surface was 1.0 mg/L decreasing to zero at 800 μm for BOD = 89 mg/L, and 2.0 mg/L decreasing to zero at 1,100 μm for BOD = 1,267 mg/L. (From Bezbaruah and Zhang (2004) *Biotechnology and Bioengineering* 88(1): 60–70. Reprinted with permission.)

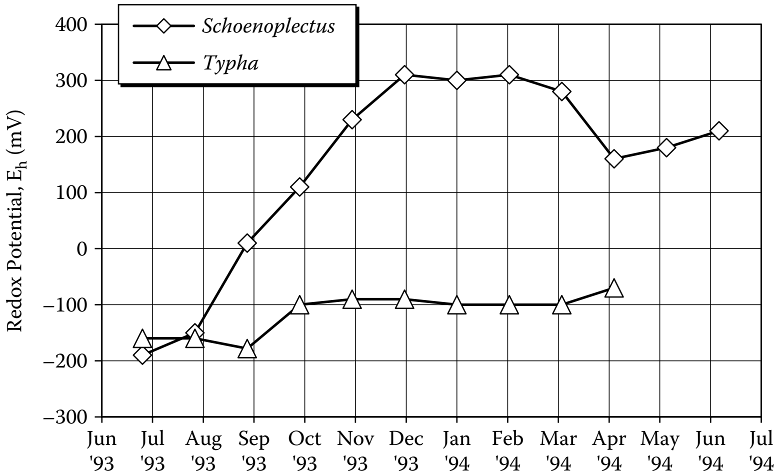


FIGURE 5.15 Annual progression of redox potential present in FWS wetland soils located in North Carolina. Data points are averages of three depths into soil (2, 5, 10 cm), three longitudinal positions (25, 50 and 75%), and two wetlands. (From Szögi *et al.* (2004) *Applied Engineering in Agriculture* 20(2): 189–200. Reprinted with permission.)

Redox Potentials in Treatment Wetlands

Szögi *et al.* (2004) studied the redox profiles in FWS wetlands receiving swine wastewater in Duplin County, North Carolina. The wetlands were of shallow depth (10 cm), and received light loadings (HLR = 2.1–2.8 cm/d, ammonia loadings 175–200 $gN/m^2 \cdot yr$). In general, there were slightly higher values near the soil surface, by 20–80 mV. The *Typha* wetlands were more anoxic than the *Schoenoplectus* wetlands (Figure 5.15).

Table 5.2 summarizes results from HSSF wetlands at Minoa, New York, and Vilagrassa, Spain. Typically, redox potentials are higher in the top layers of the HSSF beds than in the bottom. The Minoa beds were very anaerobic; the Vilagrassa beds were mildly anoxic, in terms of ORP values.

5.5 WETLAND HYDROGEN ION CONCENTRATIONS

Healthy aquatic systems can function only within a limited pH range. As a consequence, surface water discharge permits frequently require $6.5 < pH < 9.0$. Wetland water chemistry and biology are likewise affected by pH. Many treatment bacteria are not able to exist outside the range $4.0 < pH < 9.5$ (Metcalf and Eddy Inc., 1991). Denitrifiers operate best in the range $6.5 < pH < 7.5$, and nitrifiers prefer pH = 7.2 and higher. The same principles apply to other wetland biota; the acid bog vegetation is adapted to low pH, and differs greatly from the vegetation of an alkaline fen. In addition to controlling various biological processes, pH is also a determinant of several important chemical reactions. Ammonium changes to free ammonia at pH above neutral and at higher temperatures (see Chapter 9). The protonation of phosphorus changes with pH (see Chapter 10), and the hydroxide and oxyhydroxide precipitates of iron, manganese, and aluminum are pH sensitive (see Chapter 11). The pH value profoundly influences hydroxide, carbonate, sulfide, phosphate, and silicate equilibria in submerged soils. These equilibria regulate the precipitation and dissolution of solids, carbon equilibria (see last section of this Chapter), the sorption and desorption of ions, and the concentrations of nutritionally significant ions or substrates (Ponnamperuma, 1972).

Natural wetlands exhibit pH values ranging from slightly basic in alkaline fens (pH = 7–8) to quite acidic in sphagnum bogs (pH = 3–4) (Mitsch and Gosselink, 2000b). Natural freshwater marsh pH values are generally slightly acidic, (pH = 6–7). The organic substances generated within a wetland via growth, death, and decomposition cycles are the source of natural acidity. The resulting humic substances are large complex molecules with multiple carboxylate and phenolate groups. The protonated forms have a tendency to be less soluble in water, and precipitate under acidic conditions. As a consequence, wetland soil/water systems are buffered against incoming basic substances. They may be less well buffered against incoming acidic substances, since the water column contains a limited amount of soluble humics.

Treatment wetland effluent hydrogen ion concentrations are typically circumneutral. The notable exceptions are those wetlands receiving acid mine drainage, which reflect the low pH of the incoming waters. This special type of treatment wetland is not considered here; the reader is referred to Weider (1989) and Davis (1995). Furthermore, there is an important distinction between FWS and SSF systems in the ability of algae to conduct photosynthetic modulation of pH.

Surface Flow Wetlands

In aquatic systems, algal photosynthetic processes peak during the daytime hours, creating a diurnal cycle in pH. Photosynthesis utilizes carbon dioxide and produces oxygen, thereby shifting the carbonate–bicarbonate–carbon dioxide equilibria to higher pH. During nighttime hours, photosynthesis

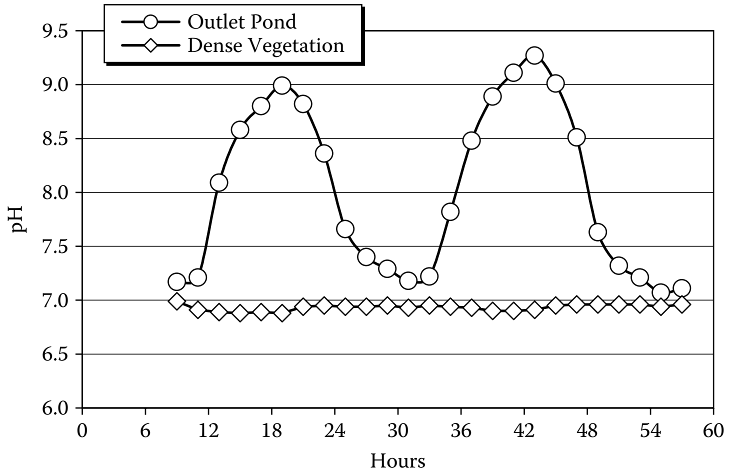


FIGURE 5.16 Diurnal variation in pH a near the exit of Cell 7 at Sacramento, California. In dense vegetation just prior to the outlet deep zone, pH does not vary. In the outlet deep zone, there is a large diurnal swing in pH, presumably driven by algal activity in the open water. (Data from Nolte and Associates (1997) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project*. 1996 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates: Sacramento, California.)

is absent, and algal respiration dominates, producing carbon dioxide and using oxygen. Open water zones within wetlands can develop high levels of algal activity, which in turn creates a high pH environment. Open water areas in wetlands also exhibit these phenomena. Diurnal pH fluctuations are not evident in areas with dense emergent vegetation. Data collected at the Sacramento, California, wetlands illustrate these phenomena (Figure 5.16). In a densely vegetated zone near the outlet, there is no diurnal cycle in pH. However, there is a large diurnal cycle in the outlet deep zone, in which the detention time is about one day. Large exports of TSS occurred episodically, indicating high algal activity, which is in turn consistent with the large pH swing.

Vegetated FWS wetlands produce effluent waters with pH just above neutrality. This occurs whether the incoming water is acidic (Figure 5.17) or basic (Figure 5.18). The Connell, Washington, wetlands treat food processing wastewater which is acidic, and which contains a large amount of nitrogen (TN of about 150 mg/L). The process of nitrification reduces alkalinity, and would be expected to drive pH downward. However, other wetland processes are involved, such as solids and COD removal, and the wetland causes a pH increase (Figure 5.17). In contrast, the Estevan, Saskatchewan, FWS wetlands treat municipal wastewater from lagoon pretreatment, which produces a high pH influent to the wetlands. The combination of wetland processes drives the pH downward (Figure 5.18).

The annual trends in FWS pH are typically quite weak (Figure 5.19). The residuals account for about one third of the variability, are normally distributed, and are independent of the time of the year. Because of these weak annual trends, FWS behavior can be adequately described by an annual mean and the associated standard deviation (Table 5.6). The pH produced in FWS treatment wetlands is within a surprisingly narrow band. Constructed systems treating municipal effluents produce an intersystem annual average of pH = 7.18 ± 0.35 (N = 20, total years data = 56). Nine of these twenty constructed wetlands exhibited a weak annual cycle, with a

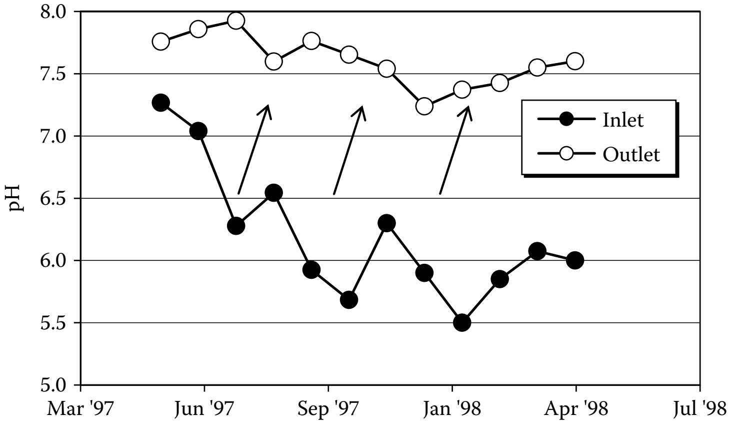


FIGURE 5.17 Water enters the Connell, Washington, FWS wetlands at low pH, and is modified to values just above neutral.

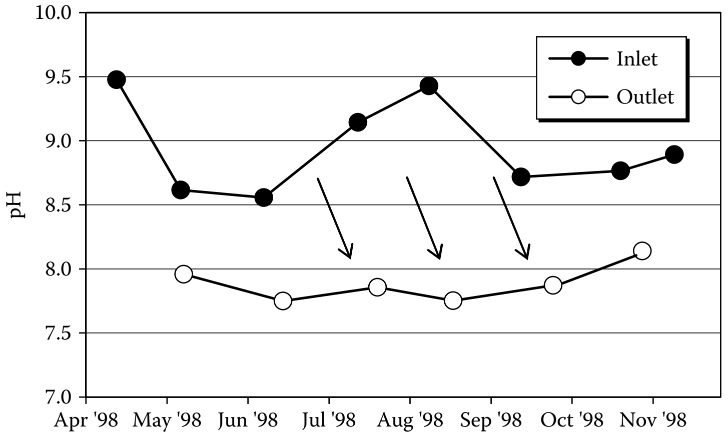


FIGURE 5.18 Water enters the Estevan, Saskatchewan, FWS wetlands at high pH, and is modified to values just above neutral.

mean amplitude of 0.25. Most of these nine contained significant amounts of open water, including terminal deep zones. Industrial and groundwater sources may cause wetlands to produce pH about a half unit higher (Table 5.6).

Natural treatment wetlands produce slightly lower pH, by about 0.5 units. This is possibly due to the antecedent peat soils that occupied most of these. Continued application of circumneutral wastewater to a naturally acidic wetland can eventually alter the pH of the surface waters in the wetland. This was the case for an acid sphagnum–black spruce bog, which received circumneutral wastewater for approximately 25 years (Kadlec and Bevis, 1990), as well as for a slightly acid peatland at Houghton Lake receiving slightly basic lagoon water. The effect on the peatland in both cases was the partial solubilization of the solid humic substances that formed under more acidic natural conditions. In addition to the chemical effect of humic solubilization, those decomposition processes that were acid-inhibited can resume under the less acidic conditions.

Treatment wetland information thus allows prediction of FWS wetland water effluent pH to within about ±0.3 units, based upon the character of the influent and the open water fraction and location in the wetland.

Subsurface Flow Wetlands

Subsurface flow wetlands also moderate and buffer the pH variations and levels of incoming basic waters (Table 5.7). There are typically weak or nonexistent annual cycles, and pH is driven to values just above neutral. For example, for the Holtby, United Kingdom, HSSF system (Figure 5.20), residuals comprise a large portion (90%) of the variability, are normally distributed, and are independent of the time of the year. Vertical, transverse and longitudinal pH profiles have been monitored at Minoa, NERCC, and Grand Lake. These data show essentially no spatial variability within the beds. As a consequence, system performance is adequately described by input/output information (Table 5.7). Twenty-four United Kingdom

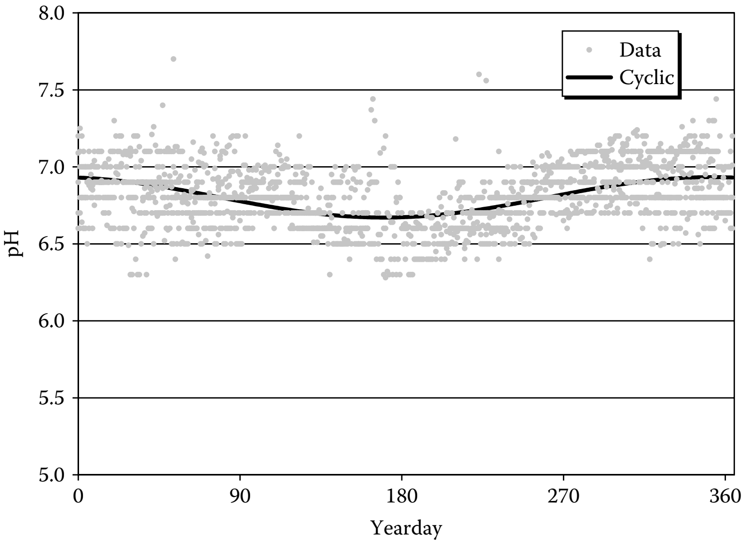


FIGURE 5.19 The annual cyclic trend in daily effluent pH from the Titusville, Florida, FWS wetland. There is a midsummer minimum, and the amplitude of the cycle is only $A = 0.13$ pH units. Trend line is a least-squares fit to an equation of the form of Equation 6.1.

TABLE 5.6
Effluent pH for Several Classes of FWS Treatment Wetlands

Site	Wetland	Location	Source Water	Data Years/ Operational Years	pH	Standard Deviation	Percent Open Water	Annual Cycle Mplitude	Summer pH	Peak Time
Constructed Municipal										
Columbia	All	Missouri	Secondary	3/10	7.41	0.12	—	None	7.41	—
Orlando Easterly	Stratum 1	Florida	Tertiary	8/13	6.91	0.19	—	None	6.91	—
Orlando Easterly	Stratum 2	Florida	Tertiary	8/13	6.87	0.20	—	None	6.87	—
Orlando Easterly	Stratum 3	Florida	Tertiary	8/13	6.99	0.21	—	None	6.99	—
Tres Rios	H1	Arizona	Secondary, partial nit-denit	2/6	7.04	0.13	25	None	7.04	—
Tres Rios	H2	Arizona	Secondary, partial nit-denit	2/6	7.06	0.10	25	None	7.06	—
Tres Rios	C1	Arizona	Secondary, partial nit-denit	2/6	7.09	0.11	15	None	7.09	—
Tres Rios	C2	Arizona	Secondary, partial nit-denit	2/6	7.12	0.10	10	None	7.12	—
Sacramento	1	California	Secondary	1/5	6.92	0.17	25	0.15	7.07	Summer peak
Sacramento	5	California	Secondary	1/5	7.06	0.15	36	0.20	7.26	Summer peak
Sacramento	7	California	Secondary	1/5	6.89	0.09	33	0.10	6.99	Summer peak
Listowel	System 3	Ontario	Lagoon	3/4	7.06	0.21	0	None	7.06	—
Listowel	System 4	Ontario	Lagoon	3/4	7.02	0.21	0	None	7.02	—
Richmond	Emergent	New South Wales	Secondary	2/3	6.78	0.20	0	0.10	6.68	Winter peak
Richmond	Open water	New South Wales	Secondary	2/3	7.83	0.61	100	0.55	8.38	Summer peak
Warangal	All	India	Screened raw	1/1	7.32	0.03	0	None	7.32	—
Byron Bay	All	New South Wales	Advanced secondary	3/3	6.91	0.31	50	0.27	7.18	Summer peak
Minot	All	North Dakota	Lagoon	1/10	7.91	0.26	59	0.29	8.20	Summer peak
Brighton	All	Ontario	Lagoon	2/3	7.57	0.21	10	0.40	7.97	Double peak
Estevan	All	Saskatchewan	Lagoon	1/6	7.84	0.22	10			—
Mean					**7.18**				**7.24**	
Standard Deviation					**0.35**				**0.45**	
Natural Municipal										
Drummond	All	Wisconsin	Lagoon	6/6	4.61	0.72	0	None	4.61	—
Houghton Lake	All	Michigan	Lagoon	14/25	6.47	0.54	0	None	6.47	—
Cannon Beach	All	Oregon	Lagoon	16/16	6.71	0.24	0	0.10	6.81	Summer peak
Genoa–Oceola	All	Michigan	RIB	11/11	6.90	0.35	5	None	6.90	—
Portage-base	All	Michigan	RIB	11/11	7.03	0.39	0	None	7.03	—
Mean (excluding Drummond)					**6.78**				**6.80**	
Standard Deviation					**0.24**				**0.24**	

Constructed Other Sources										
Des Plaines	EW3	Illinois	River	1/9	7.96	0.34	75	0.49	8.45	Spring peak
Des Plaines	EW4	Illinois	River	1/9	7.97	0.22	60	0.30	8.27	Spring peak
Des Plaines	EW5	Illinois	River	1/9	8.44	0.40	50	0.18	8.62	Spring peak
Schilling	EA	Michigan	Groundwater	3/4	7.31	0.36	20	None	7.31	—
Schilling	ET	Michigan	Groundwater	3/4	7.66	0.32	60	None	7.66	—
Schilling	WT	Michigan	Groundwater	3/4	7.54	0.38	30	None	7.54	—
Schilling	WA	Michigan	Groundwater	3/4	7.39	0.24	15	None	7.39	—
Everglades Nutrient Removal Project	All	Florida	Agricultural runoff	5/9	7.39	0.27	50	None	7.39	—
New Hanover	Raw	North Carolina	Leachate	2/4	7.70	0.38	10	0.25	7.95	Summer peak
New Hanover	Treated	North Carolina	Leachate	2/4	7.55	0.26	10	0.16	7.71	Summer peak
Connell	W1/2	Washington	Food processing	1/7	7.61	0.19	0	0.3	7.72	Summer peak
Natural Other Sources										
Northern mine	All	Ontario	Minewater lagoon	4/4	7.35	0.31	10	None	7.35	—
Mean					**7.66**				**7.78**	
Standard Deviation					**0.33**				**0.45**	

TABLE 5.7
Examples of pH in HSSF Treatment Wetlands

Site	Wetland	Location	Source Water	Data Years	Inlet pH	Standard Deviation	Outlet pH	Standard Deviation
United States								
Grand Lake	—	Minnesota	STE	4	7.33	0.28	7.16	0.19
NERCC	1	Minnesota	STE	3	7.19	0.13	7.06	0.15
NERCC	2	Minnesota	STE	3	7.19	0.13	7.06	0.16
Minoa	Planted	New York	Primary	2	7.15	0.23	7.05	0.23
Minoa	Unplanted	New York	Primary	2	7.15	0.23	7.08	0.21
Carville	—	Louisiana	Lagoon	4	—	—	7.3	0.3
Benton	—	Louisiana	Lagoon	3	8.5	0.7	7.3	0.3
Mandeville	—	Louisiana	Lagoon	1	—	—	7.2	0.2
Haughton	—	Louisiana	Lagoon	4	7.5	0.6	7.2	0.2
Benton	3	Kentucky	Lagoon	1	7.46	0.55	7.05	0.23
Australia, New Zealand								
Richmond	Cattail	NSW	Secondary	2	7.23	0.15	6.73	0.23
Richmond	Bulrush	NSW	Secondary	2	7.23	0.15	6.78	0.20
Richmond	Unplanted	NSW	Secondary	2	7.23	0.15	6.90	0.19
Portland	—	New Zealand	—	—	9.15	1.00	7.18	0.51
Waipoua	—	New Zealand	—	—	7.32	0.27	6.96	0.24
Scandinavia								
Esval	—	Norway	Leachate	5	7.5	—	7.5	—
Haugstein	—	Norway	STE	5	7.3	—	7.3	—
Tveter	—	Norway	STE	5	8.5	—	7.4	—
Mean					**7.56**		**7.12**	
Standard Deviation					**0.60**		**0.20**	

Site	Wetland	Location	Source Water	Data Years	Inlet pH	Standard Deviation	Outlet pH	Standard Deviation
United Kingdom								
Cheshire, England	1	U.K.	STE	2	7.53	0.16	7.16	0.17
Cheshire, England	2	U.K.	STE	2	7.53	0.16	7.21	0.15
Cheshire, England	3	U.K.	STE	2	7.53	0.16	7.22	0.21
Cheshire, England	4	U.K.	STE	2	7.53	0.16	7.15	0.13
Cheshire, England	5	U.K.	STE	2	7.53	0.16	7.24	0.13
Cheshire, England	6	U.K.	STE	2	7.53	0.16	7.31	0.37
Cheshire, England	7	U.K.	STE	2	7.53	0.16	7.23	0.19
Cheshire, England	8	U.K.	STE	2	7.53	0.16	8.13	0.20
Cheshire, England	9	U.K.	STE	2	7.53	0.16	7.18	0.13
Cheshire, England	10	U.K.	STE	2	7.53	0.16	7.35	0.15
Essex, England	Lower	U.K.	STE	1	8.02	0.25	7.70	0.27
Essex, England	Upper	U.K.	STE	1	8.02	0.25	7.91	0.30
Londonderry, Northern Ireland	1	U.K.	STE	7	7.02	0.25	7.09	0.56
Londonderry, Northern Ireland	2	U.K.	STE	7	7.02	0.25	7.10	0.31
Londonderry, Northern Ireland	3	U.K.	STE	7	7.01	0.26	6.98	0.27
Londonderry, Northern Ireland	4	U.K.	STE	7	6.99	0.30	6.95	0.23
Yorkshire, England	—	U.K.	STE	3	8.21	0.34	7.40	0.21
Leicestershire, England	—	U.K.	STE	2	7.54	0.35	7.50	0.19
North Yorkshire, England	—	U.K.	STE	9	7.64	0.35	7.50	0.41
Fife, Scotland	1	U.K.	STE	2	7.65	0.35	7.56	0.14
Fife, Scotland	2	U.K.	STE	2	7.65	0.35	7.88	0.11
Fife, Scotland	3	U.K.	STE	2	7.65	0.41	7.18	0.20
Fife, Scotland	4	U.K.	STE	2	7.65	0.41	7.12	0.07
Mean					**7.54**		**7.35**	
Standard Deviation					**0.31**		**0.31**	

(Continued)

TABLE 5.7 (CONTINUED)
Examples of pH in HSSF Treatment Wetlands

Site	Wetland	Location	Source Water	Data Years	Inlet pH	Standard Deviation	Outlet pH	Standard Deviation
Czech Republic								
Mořina	—	CR	Primary	2	7.76	0.21	7.69	0.10
Chlístovice	—	CR	Secondary	2	7.40	0.16	7.29	0.16
Cistá	—	CR	Primary	11	7.37	0.26	7.33	0.26
Dolní Mesto	—	CR	Primary	2	7.14	0.26	7.10	0.26
Krátká	—	CR	Primary	1	7.13	0.34	7.46	0.44
Krucemburk	—	CR	Primary	1	7.90	0.11	8.00	0.09
Ondřejov	—	CR	Primary	5	6.98	0.26	6.95	0.67
Příbraz	—	CR	Primary	7	7.04	0.47	7.17	0.46
Rudíkov	—	CR	Primary	5	7.86	0.27	7.69	0.25
Spálené Poříćí	—	CR	Primary	13	7.45	0.74	7.39	0.75
Zahrádky	—	CR	Primary	2	7.30	0.20	7.70	0.34
Žitenice	—	CR	Primary	7	7.59	0.31	7.43	0.29
Mean					**7.41**		**7.43**	
Standard Deviation					**0.31**		**0.30**	

Note: U.K. site names are approximate. STE = septic tank effluent.

reed beds had outlet pH = 7.33 ± 0.32, measured over time periods of one to nine years. However, 18 other HSSF systems located in Norway, Australia, New Zealand, and the United States had similar outlet pH = 7.12 ± 0.20, measured over time periods of one to five years. Thus, it is possible to generalize, and to expect SSF effluent pH to be just above neutrality. Also, results from the Czech Republic (Table 5.7) indicated literally no change of pH after passage through the HSSF wetlands. The average inflow and outflow pH values from the 12 systems were 7.41 ± 0.31 and 7.43 ± 0.30, respectively.

When HSSF wetlands follow a lagoon in a treatment train, algal activity in the pond often creates elevated pH entering the wetland. This may be seen for the Benton, Louisiana, system in Table 5.7. The pH modification in the wetland most likely was due to the interactions between the substrate and its biofilms, rather than to the macrophytes. Data from Richmond, New South Wales, Australia (Bavor *et al*., 1988), and from Minoa, New York (Theis and Young, 2000), support this idea, since unplanted gravel beds produced the same pH as planted systems.

Much the same conclusion may be reached for VF wetlands, which also display circumneutral pH and little or no pH change throughout the wetland (Table 5.8).

Wetlands Treating Acid Mine Drainage

There are a number of variants of constructed wetlands that target acid mine drainage, with the purpose of reducing

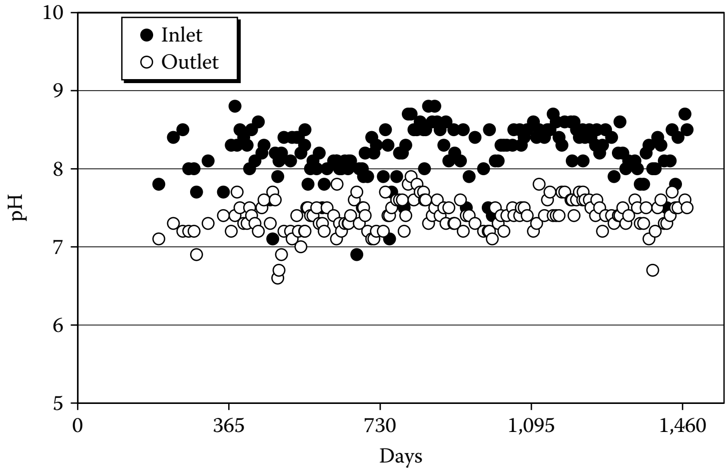


FIGURE 5.20 Inlet and outlet pH for a SSF wetland in Yorkshire, England. (Data from CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

TABLE 5.8
Hydrogen Ion in Waters Entering and Leaving Example VF Wetlands

Country	Site Name	pH In	pH Out
United Kingdom	Londonderry, Northern Ireland (gravel bed)	7.05	6.95
United Kingdom	Londonderry, Northern Ireland (peat bed)	7.03	6.82
United Kingdom	Staffordshire 1, England (1st stage vertical flow)	7.59	7.93
United Kingdom	Staffordshire 1, England (2nd stage vertical flow)	7.93	7.97
United Kingdom	Staffordshire 2, England (1st stage vertical flow)	7.39	7.81
United Kingdom	Staffordshire 2, England (2nd stage vertical flow)	7.83	7.81
United Kingdom	Somerset, England	7.70	7.70
Netherlands	Hobbitstee (Wapserveen, Netherlands)	7.47	6.88
Netherlands	Spijkerman (Wapserveen, Netherlands)	6.86	6.83
Netherlands	van Ravenhorst (Woudenberg, Netherlands)	6.59	6.91
Netherlands	van Oirschot (Boxtel, Netherlands)	7.99	7.23
Netherlands	Adema (Lemmer, Netherlands)	7.31	7.15
Netherlands	Klein Profijt (Oud Biejerland, Netherlands)	7.76	7.45
Netherlands	Nooyen (Deurne, Netherlands)	6.79	6.80
Mean		**7.30**	**7.19**
Standard Deviation		**0.45**	**0.46**

Note: U.K. site names are approximate.

Source: Data from CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom; and ECOFYT (F. van Dien).

metal content and improving (raising) the pH. However, such wetland systems typically do not change pH very much. For instance, Wieder (1989) surveyed 128 constructed wetlands, and found a mean inlet pH = 3.61 and a mean outlet pH = 3.72. However, it must be remembered that pH is the negative logarithm of concentration, and thus this small increase in pH corresponded to a median reduction in hydrogen ion concentration of 68% (Wieder, 1989). More intensive individual studies corroborate this finding. For instance, Mitsch and Wise (1998) found an inlet pH of 2.82 was increased to only pH 3.34 by passage through a FWS treatment wetland complex. Likewise, Wieder (1992) found an average increase from 2.89 to 3.08 in five organic-substrate wetlands. The frequency distribution of results of a number of other acid mine wetland studies are shown in Figure 5.21. It is clear from this body of knowledge that constructed wetlands do not provide a mechanism for adjustment of strongly acidic water conditions.

Substrate Effects

The selected substrate for both FWS and SSF wetlands can have an effect on the pH of the water, at least for a period of

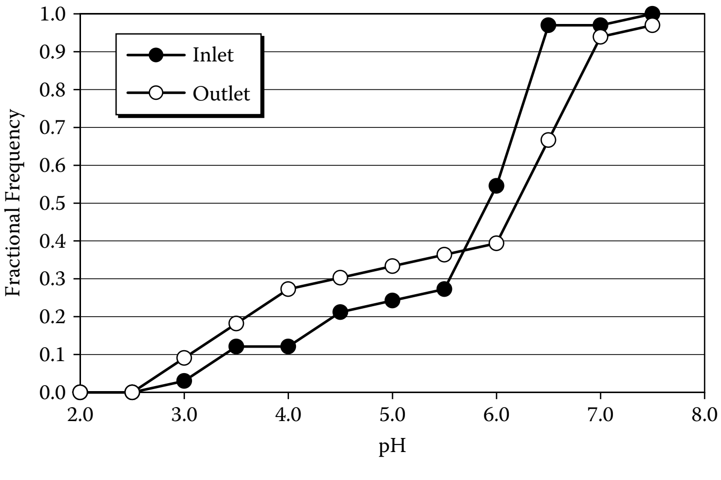


FIGURE 5.21 Distribution of pH in the inlet and outlet of acid mine drainage wetlands. (Data from 33 wetlands are represented from Brodie (1990) *Constructed wetlands for treating acid drainage at Tennessee Valley Authority Coal Facilities*. Cooper and Findlater (Eds.). Proceedings of the International Conference on the Use of Constructed Wetlands for Water Pollution Control, 24–28 September 1990, Pergamon Press: Cambridge, United Kingdom, pp. 461–470; and from Younger *et al.* (2002) *Mine Water: Hydrology, Pollution, Remediation*. Kluwer Academic Publishers, London, United Kingdom.)

time prior to alteration of the substrate. The use of alkaline ash as the substrate for a FWS wetland was investigated by Ahn *et al*. (2001). The parent ash material had pH = 10.6, and was used as a liner. First-year water pH was somewhat elevated, but after two years little or no pH effect of the liner could be found.

Mesocosm studies on the use of shale as a medium to enhance phosphorus removal in SSF wetlands were conducted by Drizo *et al*. (1997). The pH of the incoming water was approximately 7.0, but outlet waters began at pH = 4.0, and slowly rose to 6.0 over the span of 250 days of operation.

The use of zeolites or expanded clays as a medium in SSF systems poses the opposite problem: the media may be very alkaline, and create high pH in the waters being treated. For instance, Zhu *et al*. (1997) tested a variety of light weight aggregates, again for purposes of phosphorus removal. Ten different varieties of expanded clays had pH = 9.78 ± 0.53. Similarly, a number of sands tested by Brix *et al*. (2001) showed pH = 8.39 ± 0.18 (N = 13). These substrate effects may be transitory, and of small consequence in passive systems, but could be important in wetlands in which media replacement is a design intent.

5.6 ALKALINITY AND ACIDITY

Hydrogen ion content is but one possible measure of the ionic condition of a particular water. A broader concept is that of alkalinity, defined as the net concentration of strong base in excess of strong acid (Morel and Hering, 1993). Operationally, a sample is titrated with strong acid (hydrochloric or sulfuric) to an endpoint of about pH = 4.5 (APHA, 1992). In "pure" waters, the base requiring neutralization is present because of dissolved carbon dioxide and its equilibrium dissociation products:

$$\begin{aligned} CO_{2(air)} &\rightleftharpoons CO_{2(dissolved)} + H_2O \rightleftharpoons H_2CO_3 \\ &\rightleftharpoons H^+ + H_2CO_3^- \rightleftharpoons H^+ + CO_3^{2-} \end{aligned} \tag{5.22}$$

The equilibria associated with these chemical conversions are discussed in more detail in Chapter 8. The distribution of chemical species at 25°C is shown as a function of pH in Figure 8.1. The sum of all carbonate species is dissolved inorganic carbon (DIC), which is the source of energy for autotrophic microorganisms. In this simplistic pure water context, alkalinity is defined as (Pankow, 1991):

$$\text{Alkalinity} = [HCO_3^-] + 2\left[CO_3^{2-}\right] + [OH^-] - [H^+] \tag{5.23}$$

Conversion factors for the computation of DIC from pH, temperature, and alkalinity are given in Wetzel and Likens (1991). The resulting alkalinity of pure water at 25°C and pH = 8 is 50 mg/L as calcium carbonate.

Treatment wetlands are not pure waters, and the other dissolved constituents can contribute to the amount of titrating acid needed. The definition of alkalinity must then be expanded, for example to include other common components of waters to be treated (Morel and Hering, 1993):

$$\begin{aligned} \text{Alkalinity} = {} & [HCO_3^-] + 2\left[CO_3^{2-}\right] + [OH^-] \\ & - [H^+] + [NH_3] - [H_2S] + [S^{2-}] \end{aligned} \tag{5.24}$$

Phosphates, borates, and silicates may also contribute.

As seen in the preceding section, the water may be acidic rather than basic, thus requiring titration with a strong base such as sodium hydroxide. The same concepts still apply, but alkalinity is replaced by acidity (Morel and Hering, 1993):

$$\text{Acidity} = -\text{Alkalinity} \tag{5.25}$$

The concept of acidity applies particularly to wetlands treating acid mine drainage (Younger *et al*., 2002).

Alkalinity in Treatment Wetlands

Examples of alkalinity entering and leaving treatment wetlands are given in Table 5.9. In general, total alkalinity ranges upward from the values expected for pure water (approximately 100 mg/L), to much higher values for landfill leachates (>400 mg/L). FWS wetlands typically reduce alkalinity by a small margin. Conversely, HSSF systems cause a slight increase.

There are no seasonal trends of consequence. For example, regressions of total alkalinity against yearday produced essentially flat lines, with R^2 = 0.000 at Musselwhite, Ontario, 0.097 at Estevan Saskatchewan, and 0.020 and 0.037 at Tres Rios, Arizona H1 and H2.

Carbonates in Treatment Wetlands

Solid calcium carbonate, in the form of the minerals calcite and aragonite (both $CaCO_3$), may be important in the function of some treatment wetlands. In lakes, macrophytic vegetation of the littoral zone may become encrusted with massive deposits of $CaCO_3$, formed by the photosynthetic utilization of CO_2 (Wetzel, 1983). Blue-green algae growing attached to substrates also produce large deposits of carbonates (Wetzel, 1983). The precipitation of $CaCO_3$ is extremely sensitive to pH, because of the dependence of dissolved carbonate on pH. For instance, in the pure water situation, with CO_2 controlled by atmospheric equilibrium to pH = 8.3, the solubility of $CaCO_3$ is 20 mg/L calcium (Morel and Hering, 1993).

The extraction of CO_2 from an algal growth system through assimilation into algal biomass at a rate faster than it can be replaced through atmospheric CO_2 diffusion, respiration, fermentation processes, and readjustment of carbonate equilibria leads to an increase in pH level (Figure 5.17), and perhaps resulting in the precipitation of carbonates in lake environments (Wetzel, 1983).

In the wetland environment, the utilization of carbon dioxide by plants and algae also may drive the pH to high levels. The elevation in pH that results from intense SAV and periphyton photosynthesis can lead to $CaCO_3$ supersaturation, which in turn may facilitate precipitation of calcitic material. In a SAV community, submerged leaves may provide nucleating sites for $CaCO_3$ crystallization due to the very high pH levels that can occur at the leaf surface–water interface, and therefore may be important locations for encrustation with calcite.

TABLE 5.9
Total Alkalinity Entering and Leaving Example Treatment Wetlands

					Total Alkalinity		pH	
System	**Location**	**Wetland**	**WW Type**	**Years**	**Inlet**	**Outlet**	**Inlet**	**Outlet**
FWS								
Byron Bay	Australia	—	Activated Sludge	1990–1993	70	53	6.95	6.92
Orlando Easterly	Florida	WP1-MM7	Tertiary	1993–2002	94	101	7.11	6.97
Musselwhite	Ontario	—	Mine	1997–2002	133	87	7.90	6.97
Tucush	Peru	—	Mine	2006	158	148	7.51	7.95
Tres Rios	Arizona	H1	Nitrified 2°	1995–2002	174	174	6.94	7.14
Tres Rios	Arizona	H2	Nitrified 2°	1995–2002	174	181	6.95	7.22
Tres Rios	Arizona	C1	Nitrified 2°	1995–2002	176	185	6.92	7.36
Tres Rios	Arizona	C2	Nitrified 2°	1995–2002	177	184	6.92	7.23
Imperial	California	—	Ag Runoff	2001–2005	242	210	7.57	7.63
Brawley	California	—	Ag Runoff	2001–2005	278	247	7.35	7.76
Estevan	Saskatchewan	—	Lagoon	1994–2003	346	372	8.15	8.70
Champion	Florida	A & B	Lagoon	1991–1993	418	373	7.80	7.90
Champion	Florida	C & D	Lagoon	1991–1993	418	390	7.80	7.80
Isanti-Chisago	Minnesota	—	Leachate	1996–2000	480	415	7.91	7.67
New Hanover	North Carolina	—	Leachate	1997	889	536	7.85	7.84
Mean (FWS)					**282**	**244**	**7.44**	**7.54**
HSSF								
Benton	Kentucky	3	Lagoon	1988–1989	100	183	7.20	7.00
Hardin	Kentucky	1	Package	1988–1989	126	179	7.00	7.10
Hardin	Kentucky	2	Package	1988–1989	126	164	7.00	7.20
Grand Lake	Minnesota	—	Septic Tank Effluent	1995–1998	383	393	7.23	7.12
NERCC	Minnesota	1	Septic Tank Effluent	1996–1999	466	479	7.23	7.07
NERCC	Minnesota	2	Septic Tank Effluent	1996–1999	466	488	7.23	7.08
Mean (HSSF)					**278**	**314**	**7.15**	**7.10**

This carbonate chemistry has extremely important ramifications in the south Florida environment. Carbonates dominate the substrates of some natural Everglades systems, notably the marl prairies (Gleason and Stone, 1994) (Figure 5.22). Phosphorus coprecipitates with the calcitic solids, and therefore there have been attempts to emulate the natural system with constructed wetlands designed to remove phosphorus. There are two principal variants on the theme: systems that maximize algal components and the availability of calcium substrates (periphyton systems), and those that maximize submerged plant surfaces and their photosynthesis (SAV systems). Periphyton system data has been

FIGURE 5.22 Calcium carbonate deposits supported by the canopy of sawgrass (*Cladium jamaicense*) in the marl prairie of the Florida Everglades. These form as the result of calcitic periphyton mats that desiccate during dry periods.

reviewed and summarized in Kadlec and Walker (2004). The SAV systems use the structure of the underwater plants to support algal encrustations (Dierberg *et al*., 2002). Limerock components may be added to the SAV system to augment the calcium supply (DeBusk and Dierberg, 1999). At the time of this writing, large-scale (40 ha each) demonstration projects are underway at two locations. One site was prepared by scraping all soil from a limerock base. The second was prepared by incorporating calcareous material in the top layer of a sand bed material. No results are available at the time of this writing.

The purpose of treatment in acid mine wetlands is the removal of acidity, or equivalently, the production of alkalinity. The terms *reducing and alkalinity producing systems* (RAPS) and *successive alkalinity producing systems (*SAPS) have been coined to describe wetland systems designed for that purpose (Younger *et al*., 2002). These are beds of organic material overlying limestone rock beds, through which the water flows downward. The depth of standing water on top of these beds determines the type and vitality of the wetland plants. This concept is further discussed in Chapter 11 (see Figure 11.15).

SUMMARY

Some natural wetlands operate with high levels of dissolved oxygen, others at low levels. Most treatment wetlands receive enough BOD and NOD loading to drive the DO level down to about 1–2 mg/L. Exceptions are oversized wetlands receiving very clean effluents. The soils, sediments, and biota in the wetland exert a strong influence on the DO concentrations in the water. Therefore, it is not accurate to assume that BOD and NOD disappearance is a measure of oxygen transfer.

In the case of FWS wetlands, most O_2 transfer is probably due to interfacial aeration and underwater photosynthetic production. In HSSF wetlands, interfacial aeration is significantly reduced because the water surface is not exposed above the bed media, and also because the bed media increases diffusional distances. Therefore it is not surprising that HSSF wetlands exhibit extremely low oxygen transfer rates, which can be exceeded by oxygen demands exerted by internal wetland processes.

Based on the current body of knowledge, there is little or no evidence that FWS and HSSF wetlands are inherently "aerobic" and will automatically produce high effluent DO levels. VF wetlands that rely on principles of unsaturated flow (see Chapter 2) and design variants of SSF wetlands such as fill-and-drain (tidal flow) and aerated systems may achieve this standard, at the cost of additional mechanical input (see Part II of this book).

There is a growing body of evidence that plants provide no significant aeration flux to the water or soil, in excess of their respiratory demands.

In sediments and submerged soils, redox potential ranges from around −400 mV (strongly reduced) to +700 mV (well oxidized) and is better poised and fairly reproducible at the more reduced levels. Redox potentials are strongly influenced by the influent carbonaceous and nitrogenous oxygen loadings, internal oxygen demands within the wetland, and the rate of oxygen transfer into the wetland.

Treatment wetlands typically operate at circumneutral pH for influents that are not strong acids or bases. This is true for both FWS and SSF constructed wetlands. One exception is for wetlands designed to treat acid mine drainage, where low influent pH levels are the norm. Some HSSF wetlands are designed with reactive medias in the bed material (zeolite, LECA, or blast furnace slag; discussed in further detail in Chapter 10) that can produce high pH effluents. Unless reactive medias are employed (or the influent is highly acidic), effluent DO, pH, redox, and alkalinity levels from treatment wetlands are typically driven by carbonate equilibrium chemistry dictated by the air–water interface.

6 Representing Treatment Performance

This chapter examines the available means of collecting and analyzing the large amount of performance data that now exists for treatment wetlands. Wetlands are "open" systems heavily influenced by environmental factors. This makes them more complex than other types of biological treatment reactors (activated sludge, trickling filters) described in the environmental engineering literature. Nevertheless, attempts have been made to adapt models from these other technologies to treatment wetlands (Burgoon *et al*., 1999; McBride and Tanner, 2000; Langergraber, 2001; Rousseau *et al*., 2005b; Wu and Huang, 2006). Wetlands are dominated by biomass storage compartments that are very large relative to pollutant mass in the water column (again, different than other biological reactors). These biomass storage compartments are affected by seasonal cycles that are different than temperature cycles.

Treatment performance is represented by two components: the central treatment tendency for a wetland (or group of wetlands) and the anticipated variability away from that central tendency. Central tendencies are driven by flows and concentrations, in concert with environmental factors. Random events within the wetland will produce stochastic variations in effluent performance. Both must be assessed to describe treatment performance in constructed wetlands.

Different types of wetlands (e.g., wetland configuration, vegetative community) function differently. Therefore, a set of "universal" parameters for describing treatment performance in wetlands is not to be expected.

6.1 VARIABILITY IN TREATMENT WETLANDS

Two types of variability are of interest for understanding and design of treatment wetlands. First, it is necessary to understand the scatter of performances for an individual wetland, around either the central tendency of data or the model characterization of that central tendency. This is the intrasystem, or internal variability, and it is needed to understand the excursions that may be expected, and to design to meet permit requirements that involve allowable maximums. Internal variability includes seasonal, stochastic, and year-to-year changes. Wetland performance can also change from year to year due to changes in vegetative communities, hydraulic or organic loadings, or weather conditions. Second, it is useful to understand how comparable wetlands vary, which is the intersystem variability. Causes of this variation will include factors such as vegetation species, system geometry, and climatic conditions. Both types of variability are best explored by graphical methods.

Intrasystem Variability

Data frequency influences the degree of scatter in data. Variability decreases daily–weekly–monthly–annual, but the central tendency is the same. For example, the coefficients of variation for total phosphorus over four years at Brighton, Ontario, were weekly = 89%, monthly = 83%, and annual = 19%.

Many factors contribute to random variability in the outlet concentrations from a single treatment wetland. This variability is typically not small, with coefficients of variation of 20%–60% being common. Deterministic models reproduce the central tendency of performance, but not the random variability. Whether there is microbial or vegetative control, seasonal patterns of wetland variables are the rule, accompanied by a random variable term (Kadlec, 1999a).

Data Folding

A choice may be made to either deal with "raw" data or detrend a concentration time series using either a mechanistic model or a cyclic annual trend. Most of the existing treatment wetland literature considers the probability distributions of the raw data for concentration time series. The typical method is to present the cumulative probability distributions for concentrations entering and leaving the wetland (see, e.g., Kadlec and Knight, 1996; U.S. EPA, 1999). Typical probability distributions are shown for weekly average for data Columbia, Missouri (Figure 6.1). The median inlet BOD = 26 mg/L in 1995, while the median outlet BOD = 9 mg/L. However, inlet concentrations ranged from 8 to 60 mg/L, and outlets from 4 to 24 mg/L. At the weekly time scale, the maximum BOD exiting the wetland was 2.7 times the median. The data in this BOD example are not detrended.

Seasonal changes in treatment performance can often be represented by cosine trends (Kadlec, 1999a).

Stochastic variability will report as a "cloud" around the seasonal trend line:

$$C = C_{avg}\left[1 + A \cdot \cos\left(\omega(t - t_{max})\right)\right] + E \qquad (6.1)$$

where

A = fractional amplitude of the seasonal cycle, dimensionless
C = instantaneous outlet concentration, mg/L
C_{avg} = average (trend) outlet concentration, mg/L
E = random portion of the outlet concentration, mg/L
t = time of the year, Julian day
t_{max} = time of the year for the maximum outlet concentration, Julian day

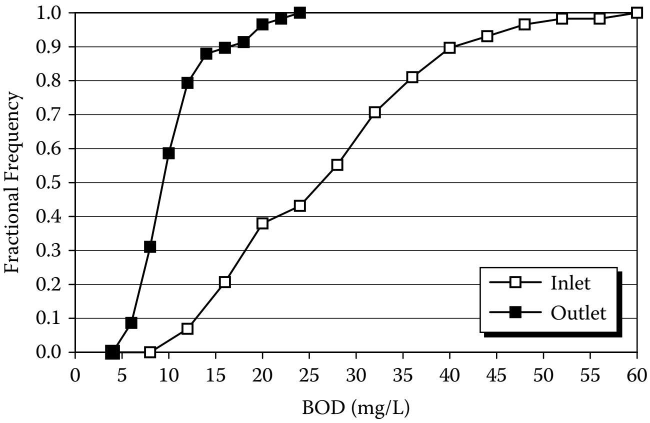


FIGURE 6.1 Distribution of BOD concentrations measured at the Columbia, Missouri, FWS wetlands in 1995. (Unpublished data from city of Columbia.)

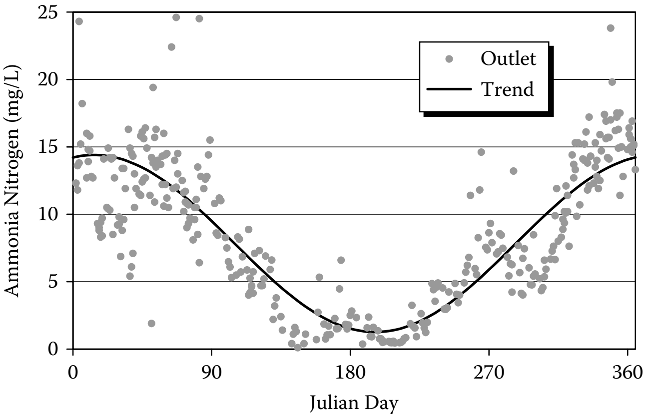


FIGURE 6.2 Ammonia nitrogen concentrations leaving the Columbia, Missouri, FWS wetlands in 1995, together with the annual trend. Data were acquired daily on weekdays. (Unpublished data from city of Columbia.)

The deterministic portion of this representation may in turn be modeled by the *k*-rate technique with appropriate rate constants and background concentrations, both of which may respond to temperature and season, as will further be discussed.

The existence of the error term (E) means that sampling must either be at high frequency or cover many annual cycles before meaningful trend averages can be determined. Data from several years may be "folded" to create an annualized grouping, distributed across the year according to Julian day. This use of many annual cycles has the advantage of including year-to-year variations in climate, flow, and ecosystem condition.

The stochastic portion (E) will have a probability distribution, which will be different depending upon sampling frequency and sample averaging period. The ammonia concentration data for Columbia, Missouri, serve to illustrate that stochastic variability may be considered separately from annual trends. At that site and most other treatment wetlands, there is a strong annual cycle in ammonia, occasioned by the slow-down in treatment during the winter months, as well as by trends in the ammonia levels leaving pretreatment (Figure 6.2). For that FWS system, Equation 6.1 was calibrated to the data from 1994 to 1995 as follows:

$$\text{Inlet:}\quad C_{avg} = 10.0 \qquad A = 0.61 \qquad t_{max} = 19$$

$$\text{Outlet:}\quad C_{avg} = 7.8 \qquad A = 0.84 \qquad t_{max} = 14$$

The variability in the inlet and outlet concentrations may then be expressed as fractional departures from the trend values, which is the random variable denoted by E/C from Equation 6.1 The cumulative probability distributions for both inlet and outlet time series are similar (Figure 6.3).

Intersystem Variability

Apart from the concept of how one wetland may vary in its performance, there is the issue of how the parameters of the deterministic portion of the wetland performance model change from system to system. Typically, the difference in treatment performance between wetland systems is much greater than the difference in performance within a particular wetland system. There are several ways to express this variability, including:

- Side-by-side comparisons of wetlands with different attributes, such as type, or presence, or absence of vegetation
- Distributions of model parameter values, such as *k*-values, across a large number of comparable wetlands
- Graphical performance comparisons for sets of wetlands, based upon some period of performance such as annual or entire period of data record

The key to assigning differences to "variability" is the process of accounting for the principal factors affecting performance

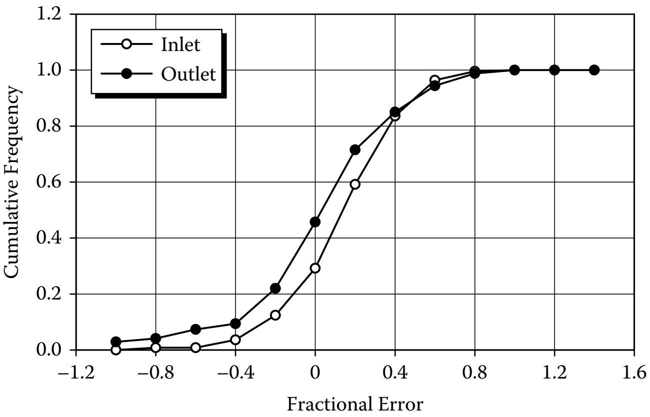


FIGURE 6.3 Distribution of ammonia concentration fractional departures from annual trends measured at the Columbia, Missouri, FWS wetlands in 1995. Derived from the data in Figure 6.2.

separately and in advance of comparison. For example, the methods for describing effects of detention time or hydraulic loading, inlet concentration, temperature, and season will be discussed in the following text. It is clear that it is not useful to compare the summer behavior of one wetland to the winter behavior of another, because we have already identified the potential for seasonal and temperature differences. A choice that minimizes seasonal effects is the annual averaging period, which retains climatological effects, such as mean annual temperature and rainfall.

Replication

Two wetlands of the same size and type should be expected to perform similarly if they receive identical water flows and concentrations. This has generally been observed to be the case in the few side-by-side studies that have involved such replication (see, e.g., Moore *et al.*, 1994; CH2M Hill and Payne Engineering, 1997; CH2M Hill, 1998; Mitsch *et al.*, 2004). Typical effluent concentration patterns follow similar time series, with occasional differences of unknown causes (Figure 6.4). Because of the expense of building and monitoring replicated wetlands, most of the comparative studies of treatment wetlands have not involved replication; this is apparently a justifiable step.

Side-by-Side Studies

There have been numerous side-by-side studies conducted to elucidate possible effects of vegetation type, media size, aspect ratio, and other factors. In general, such studies have not involved replication, as noted in the previous text. In these studies, the incoming water chemistry and often the inlet flow rates are the same. Climatological effects, such as rainfall and air temperature, are identical for the comparison systems. The results of side-by-side testing determine the effect of the tested variable, but only for the specific circumstances of test wetland systems. For instance, Wolverton *et al.* (1983) bench-tested *Phragmites* and bulrushes (*Schoenoplectus* (*Scirpus*) spp.) in HSSF wetland microcosms and determined a significantly better performance for *Phragmites*. On the other hand, Gersberg *et al.* (1986) tested *Phragmites* and bulrushes (*Schoenoplectus* (*Scirpus*) spp.) in outdoor pilot HSSF wetland environments at Santee, California, and determined a significantly better performance for *Schoenoplectus*. However, when the same plants were tested in a full-scale HSSF facility at Minoa, New York (Liebowitz *et al.*, 2000), essentially no difference was found for COD and other parameters (Figure 6.5). These analyses emphasize the need for great care in detailing the circumstances of side-by-side studies. Further extrapolations to other situations may be very misleading, however similar the circumstances may be.

Aggregated Data Sets

Combining performance data from different wetland systems to create an aggregated data set results in data clouds that have considerably more variability than the individual wetland data sets they were created from. These aggregated data sets are useful for exploring the bounds of treatment performance in a particular application, but may not accurately predict the performance of an individual treatment wetland.

Aggregated data sets can be used to define the central tendency in treatment performance for a given type of wetland reactor and application (e.g., BOD removal in HSSF wetlands). However, use of the central tendency to create a "rule of thumb" is only one piece of the description of treatment performance. Because of the loss of specificity and high variance in these aggregated data sets, statistics such as confidence intervals and effluent multipliers have to be developed to assess short-term variances that may be important for risk assessment.

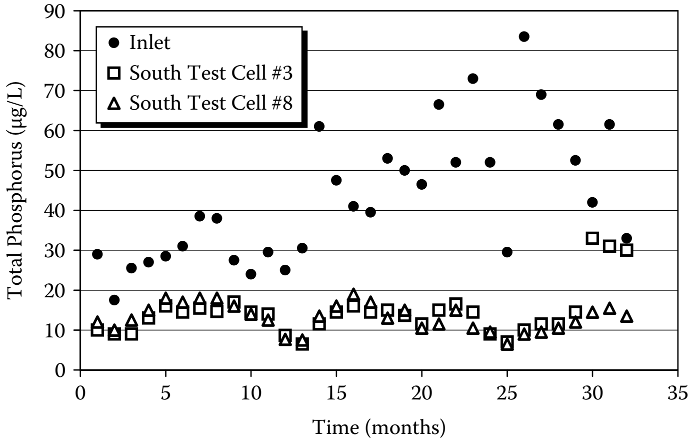


FIGURE 6.4 Performance of two FWS wetland replicates for phosphorus reduction at low concentrations. These behave similarly over most of the period of record. The reason for departure during the last three months of the record is not known. (Unpublished data from South Florida Water Management District.)

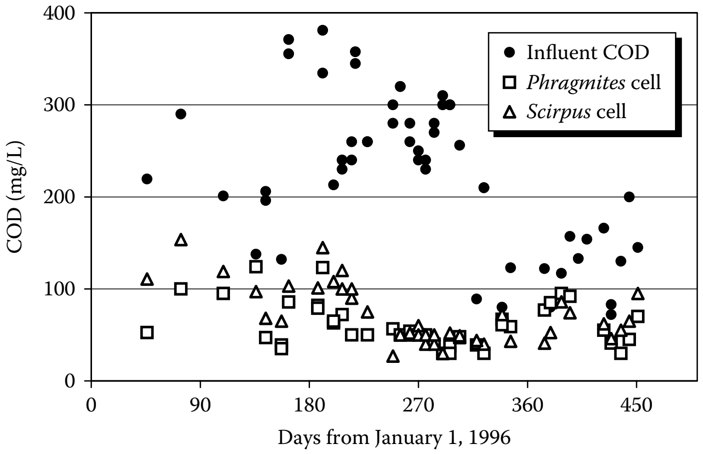


FIGURE 6.5 Performance of side-by-side wetlands at Minoa, New York, vegetated with *Phragmites* spp. and *Scirpus* (*Schoenoplectus*) spp. (Data from Theis and Young (2000) *Subsurface flow wetland for wastewater treatment at Minoa*. Final Report to the New York State Energy Research and Development Authority, Albany, New York.)

6.2 GRAPHICAL REPRESENTATIONS OF TREATMENT PERFORMANCE

There exist a large number of data sets for some of the more common pollutants, such as TSS, BOD, phosphorus, and nitrogen species. Several types of graphs may be used to compare performances across systems, and these have been used in prior treatment wetland literature:

1. Output concentration versus input concentration
2. Output concentration versus input areal loading
3. Output loading versus input loading
4. Load removed versus input areal loading
5. Rate constant versus input areal loading

The first two of these are useful representations, but the last three very often lead to spurious relationships that serve no useful purpose. Many important variables are lost in these plots, because of their restrictive 2-D nature.

Outputs Versus Inputs

The input–output concentration graph essentially extends the idea of percent removal to a group of wetlands. That is useful in obtaining first estimates of the potential of a class of treatment wetlands to reduce a particular contaminant. But, that plot is of no value in sizing the wetland, because it does not contain any information on the detention time or hydraulic loading.

The phosphorus concentration produced in treatment wetlands depends upon three primary variables (area, water flow, and inlet concentration), as well as numerous secondary variables (vegetation type, internal hydraulics, depth, event patterns, and others). It is presumed that the area effect may be combined with flow as the hydraulic loading rate (q = HLR) since two side-by-side wetlands with double the flow should produce the same result as one wetland. Therefore, two primary variables are often considered: HLR and inlet concentration (C_i). Older kinetic removal models (e.g., the k-C^* model) and performance regressions are based upon these two variables (Kadlec and Knight, 1996).

Later in this chapter, it will be shown that wetland outlet concentrations are often well represented by:

$$\frac{C_o - C^*}{C_i - C^*} = \frac{1}{(1 + k/Pq)^P} \tag{6.2}$$

where

C_o = outlet concentration, mg/L
C_i = inlet concentration, mg/L
C^* = background concentration, mg/L
k = modified first order areal constant, m/d
P = apparent number of TIS
q = hydraulic loading rate, m/d

Here this model is used to explore the expected corresponding appearance of intersystem performance graphs.

An equivalent approach is to rearrange the primary variables, without loss of generality, by using inlet loading rate (LRI = $q{\cdot}C_i$) and concentration (C_i). Thus, it is expected that the effluent concentration produced (C_o) will depend upon LRI and C_i. A graphical display has often been adopted in the literature (Kadlec and Knight, 1996; U.S. EPA, 2000a). In the broad context, multiple data sets are represented by trends that show decreasing C_o with decreasing LRI, with a different trend line associated with each inlet concentration (Figure 6.6). For low inlet concentration or for higher hydraulic loadings, the log–log slope of the data cloud is approximately 0.33 (Figure 6.6), but the resultant outlet concentration range moves upward to higher values. The right-hand asymptote of each data group, at very high pollutant loading, is an outlet concentration equal to the inlet concentration—or in other words, no removal. The left-hand asymptote, reached only for low inlet concentrations, is the background concentration, C^*. The fact that there exist data clusters for each inlet range indicates that there are at least two major factors influencing outlet concentration: inlet concentration and inlet loading.

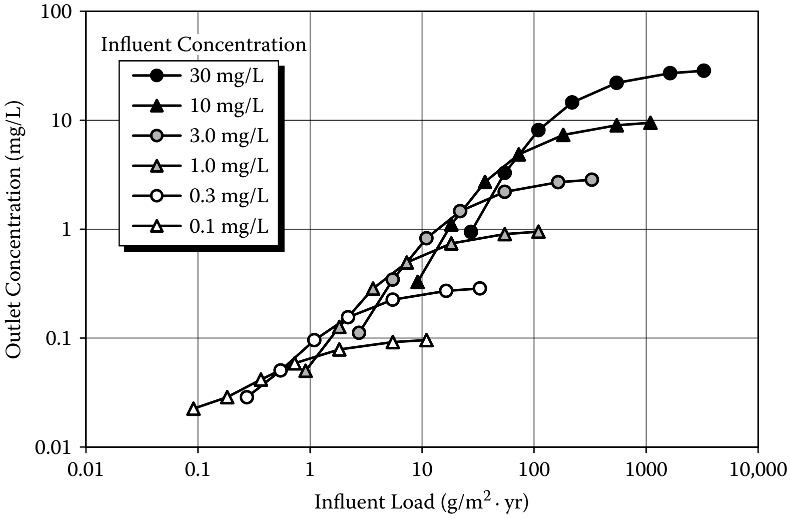


FIGURE 6.6 Hypothetical concentration load response for the *P-k-C** model, with $P = 3$, $k = 6$ m/yr, and $C^* = 0.02$ mg/L. The lines are for different values of influent concentration, as indicated in the legend. On each line, the hydraulic loadings are from left to right: 0.25, 0.50, 1.0, 2.0, 5.0, 15.0, and 30.0 cm/d.

If the entire set of points in Figure 6.6 is considered, ignoring the effect of inlet concentration, the general trend line has a log–log slope of about 1.0. However, such a single variable plot is nonunique, because of the effect of inlet concentration, and may be misleading. For instance, use of a small intersystem data set might result in use of left data points for high C_i, as well as right data points for low C_i, thus exaggerating the slope. Consequently, the C_o– LRI graph advocated in some literature (U.S. EPA, 2000a) is inadequate. The *P-k-C** model typically spans the entire cloud of intersystem results when exercised for various choices of C_i, k, and C^* (Kadlec, 1999c). It is expected that real data would display behavior like that in Figure 6.6, and that expectation is found to be realized in later chapters concerning individual contaminants.

The outlet concentration load graph is a useful addition to the design sizing toolkit for treatment wetlands. However, it cannot be used in isolation as a design sizing basis, because it does not separate the individual effects of inlet concentration and hydraulic loading. Inspection of Figure 6.6 shows that the inlet loading is not a unique design variable, and that the hydraulic loading and inlet concentration that define it are not interchangeable. Part II of this book discusses the use of a concentration-loading graph as an important component of the design process.

Perspectives Derived from the Loading Graph

The principal tool or examination of intersystem variability in this book will be the outlet concentration versus inlet loading graph. The period of data averaging involved for comparison purposes should be long enough to encompass as much as possible of the intrasystem or internal variability, so as to focus on system differences. The operations of the systems being compared should be past start-up, so that sustainable performance can be analyzed. A subtle paradox occurs due to the fact that periods of record will not typically be equivalent among comparison wetlands, except in side-by-side studies. Suppose Wetland *A* has two years and Wetland *B* has ten years, respectively, of data past start-up. Neither Wetland *A* nor Wetland *B* will necessarily operate or perform in the same way from year to year, so the choice of annual averaging will produce two distinct data points for Wetland *A* and ten for Wetland *B*. There will be interannual variability represented for each, which will, to some extent, obfuscate the comparison between Wetlands *A* and *B*. Thus there are two logical choices: the use of interannual, intersystem information, involving one point for each year for each wetland; and the use of period of record (POR), intersystem data, involving one point for each wetland.

These concepts are illustrated in Figure 6.7 for phosphorus reduction for two similar wetlands treating facultative lagoon effluents. Brighton provides some phosphorus removal via alum pretreatment, with a long-term mean influent of 0.45 mg/L. In contrast, the inlet to the Estevan (Saskatchewan) wetlands was 2.26 mg/L. Removal was 24% at Estevan, at an average hydraulic loading of 2.6 cm/d over a nine-year period of record past start-up. Removal was 40% at Brighton, at an average hydraulic loading of 5.1 cm/d over a 4.25-year period of record (POR) past start-up. Data are shown as monthly, annual, and period of record averages of weekly measurements. The monthly data scatter is in part due to seasonal differences, which spanned May through November for Estevan, and all 12 months for Brighton. This seasonal effect is removed by annual averaging, which causes only interannual and intersystem effects to remain. Finally, interannual effects are removed by constructing the period of record averages, involving four years for Brighton and nine years for Estevan.

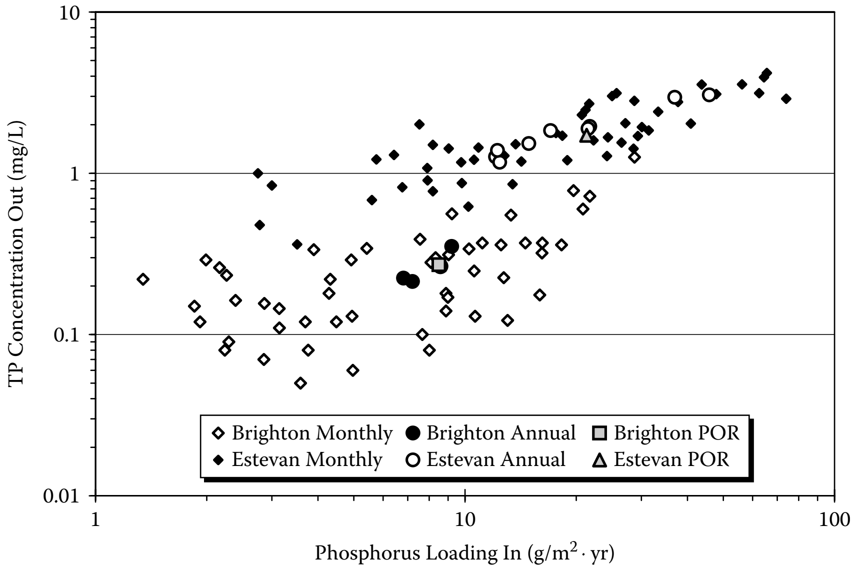


FIGURE 6.7 Outlet TP concentration versus inlet TP loading for Estevan, Saskatchewan, and Brighton, Ontario, treatment wetland systems. The period of record past start-up was 4.25 years for Brighton, and nine years for Estevan. (Unpublished data from city of Estevan and city of Brighton.)

The reasons for the differences between these two systems cannot be determined from the graphical representation. However, as shall be seen in Chapter 10, much of the difference is attributable to the nonuniqueness of the phosphorus-loading variable, meaning that the difference in inlet concentrations places the two systems in different groupings.

It is also possible to look further via the *P-k-C** model. There are no tracer tests of either wetland, so it will be presumed that $N = P = 4$. It is known that C^* is quite low for phosphorus, and it will be presumed that $C^* = 0.01$ mg/L. The POR data then indicate an annual $k = 11$ m/yr for Brighton, and $k = 3$ m/yr for Estevan.

Pitfalls of Graphical Representations

The purpose here is to illustrate the fallacy of graphical data representations and associated regressions between variables that contain the same multiplier and the errors that accompany an incorrect model choice. This subject has been elucidated for natural treatment systems by Von Sperling (1999). As a hypothetical example, consider concentrations entering the wetland vary randomly between 0.2 and 1.2 g/m³. Likewise, the concentrations leaving are also random between 0.1 and 0.3 g/m³. Therefore, the mean inlet concentration is 0.7 g/m³, the mean outlet concentration is 0.2 g/m³, and the resulting average concentration reduction is 71%.

A set of 50 experiments is run, in which the hydraulic loading is varied linearly between 1 and 50 m/yr. For any experiment, the inlet and outlet concentrations are independently random within the ranges selected (Figure 6.8). Not surprisingly, linear regression of the input/output concentrations explains virtually none of the variability There is a 72 ± 18% (mean ± SD) concentration reduction, and that is all that may be determined.

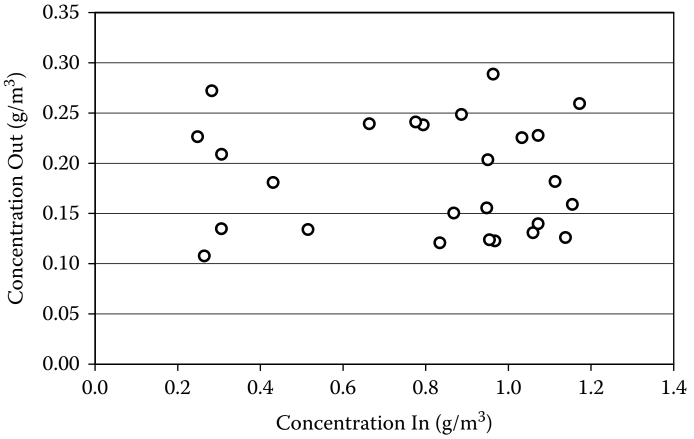


FIGURE 6.8 Scatter plot of input and output concentrations for a hypothetical data set for 71% reduction.

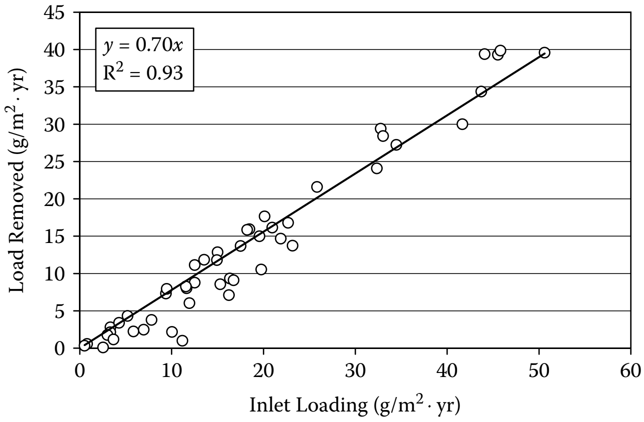


FIGURE 6.9 Load reduction versus incoming load for the hypothetical, random data set of Figure 6.8.

Next, the correlation between pollutant load reduction and inlet pollutant loading is examined. Pollutant loading is defined as hydraulic loading multiplied by concentration, for both the inlet and outlet. Pollutant load reduction is the difference between inlet and outlet pollutant loadings. A wonderful correlation is obtained with an $R^2 = 0.93$, which makes the data look great and makes us feel that we can use this for design (Figure 6.9). Unfortunately, there is no connection of performance to inlet loading, no matter how much this load graph appeals to us. The hydraulic loading appears in both the ordinate and the abcissa, thus causing a stretching of a random 2-D cloud along a diagonal axis. The only useful feature of the graph is the slope of the line, 0.70, which is the correct result for the percent reduction. Many examples of this representation and analysis are to be found in the treatment wetland literature (Knight *et al.*, 1993; Hammer and Knight, 1994; Vymazal, 2001), but they are of virtually no value in design.

The formerly popular first-order plug-flow model is then examined. The same hypothetical random data set is easily manipulated to calculate a k-value for each pair of input–output concentrations, or to provide a least-squares estimate that best fits the entire data set, according to:

$$k = q \cdot \ln\left(\frac{C_i}{C_o}\right) \tag{6.3}$$

The k-values so calculated average 32 m/yr. The important question is whether this model fits the data, so that it may be used for predictions at specified hydraulic loading rates. The answer is that the first-order model fails and predicted concentrations scatter randomly with respect to observed concentrations.

The subtle trap that has created trouble, in this example and in some of the existing treatment wetland literature, is the failure to check whether or not the model has any validity. That can be done in a number of ways, but the easiest method is the direct examination of the data trends expected from the model. For the simple first-order case, the fraction of pollutant remaining is expected to decline exponentially with detention time, or equivalently with the inverse of hydraulic loading, as indicated by Equation 6.3. In the present hypothetical example, log-linear regression of data in this manner has an $R^2 = 0.000$.

6.3 MASS BALANCES

There are many measures and models for pollutant reductions in treatment wetlands. In this chapter, various definitions and options for system description are explored as a necessary precursor to the discussions of individual pollutants that follow in ensuing chapters.

CONCENTRATIONS

Individual concentration measurements are very often averaged to eliminate some of the variability inherent in wetlands. The time average concentration, denoted by an overbar ($\bar{C}$), is defined as

$$\bar{C} = \frac{1}{t_m}\int_0^{t_m} C\,dt \tag{6.4}$$

where
C = chemical concentration, mg/L
t = time, d
t_m = averaging period, d

Such average concentrations may be acquired from time-proportional autosamplers, or computed from a time series.

An average mass flow of a chemical ($\overline{QC}$) is the product of the average flow and the flow-weighted (or mass average) concentration, defined by:

$$\hat{C} = \frac{\frac{1}{t_m}\int_0^{t_m} QC\,dt}{\frac{1}{t_m}\int_0^{t_m} Q\,dt} = \frac{1}{\bar{Q}t_m}\int_0^{t_m} QC\,dt \tag{6.5}$$

where the "hat" notation indicates a flow-weighted average.

$$\overline{QC} = \bar{Q}\,\hat{C} \tag{6.6}$$

Percent concentration reduction is often used in the literature:

$$\%\text{ Concentration reduction} = 100 \times \left(\frac{C_i - C_o}{C_i}\right) \tag{6.7}$$

This term is quite ambiguous, because it usually refers to the average of one or more synchronous samples for selected stream flows. Such contemporaneous measures do not properly reflect the internal chemical dynamics of the wetland, such as production of the chemical. Further, dilution or concentration due to rain (ET) or other unaccounted flows renders this an imperfect measure of true removal. Nevertheless, this terminology is frequently used in the literature.

Chemical Terminology

It is important to distinguish among the various measures of global wetland chemical removal. Some further definitions used in this book are specified in the following text.

Inlet Mass Loading Rates

$$M_i = Q_i C_i \tag{6.8}$$

$$m_i = \frac{M_i}{A} = \frac{Q_i C_i}{A} = q_i C_i \tag{6.9}$$

where
M_i = inlet mass loading, g/d
m_i = (specific) inlet mass loading, g/m^2·d

Acronyms are also often used for designating the chemical; for example, PLR denotes *phosphorus loading rate*. A chemical loading rate is a measure of the distributed "rainfall" equivalent of a chemical mass flow. It does not imply the physical distribution of water uniformly over the wetland.

Mass Removal Rate

$$\tilde{J}A = (Q_i C_i - Q_o C_o) \tag{6.10}$$

This represents the areal average amount of a chemical that gets stored, destroyed, or transformed. This single-number measure of wetland performance can be misleading in the common event of strong concentration gradients and removal gradients.

Percent Mass Removal

This quantity links water losses and gains to chemical losses and gains.

$$\%\text{ Mass removal} = 100 \times \left(\frac{Q_i C_i - Q_o C_o}{Q_i C_i} \right) = 100 \times \left(\frac{m_i - m_o}{m_i} \right) \tag{6.11}$$

$$\left(1 - \frac{\%M}{100}\right) = \left(1 - \frac{\%Q}{100}\right)\left(1 - \frac{\%C}{100}\right) \tag{6.12}$$

where
$\%C$ = percent concentration reduction
$\%M$ = percent mass removal
$\%Q$ = percent flow reduction

The term is less ambiguous than concentration reduction, because it traces the chemical of interest, and accounts for the effect of the quantity of water in which that chemical is located. However, the difficulties of contemporaneous measurement remain.

The Utility of Reduction Numbers

It is very easy to compare the amounts of a pollutant in the inlet and outlet streams of a wetland, and to compute the percentage difference. Unfortunately, this information is of very limited use in design or in performance predictions, because it reflects none of the features of the ecosystem, which are the target of design. By implication, it would be necessary to replicate the wetland that produced the percentage data, as well as to replicate the operating and environmental conditions that prevailed during data acquisition. The second is clearly impossible, and past experience has given strong indications that the first is also difficult.

The literature is replete with review papers that tabulate removals for a selected spectrum of wetlands (e.g., Strecker *et al.*, 1992; Cueto, 1993; Johnston, 1993).

The implication is that wetlands of a similar type will achieve a similar reduction. Whereas such groups of data begin to elucidate the bounds of performance, the effects of size, loading, flow patterns, depth, and other design variables cannot be deduced from efficiency values alone.

In some instances, the incoming concentration of a particular chemical may be small for some period of time. Then, due to measurement errors or small transfers from wetland, storages and productions may give outflow concentrations that are greater than the incoming values. A one-time calculation of a "reduction efficiency" will properly reflect that condition as a (large) negative percent reduction. At other times, a larger inflow concentration may be reduced by the wetland, leading to a positive percentage removal. If the removal percentages are then averaged, the large negative value improperly dominates the calculation.

As a result of these considerations, great care must be exercised in interpretation of percentage reduction values.

Chemical Mass Balances

Measurements of chemical composition of wetland inflows and outflows are the most obvious method of characterizing water quality functions. However, such measurements by themselves can be very misleading. A much better characterization is achieved by computing the mass balance or budget for an individual chemical constituent.

A proper mass balance must satisfy the following conditions:

1. The *system* for the mass balance must be defined carefully. A system in this context means a defined volume in space; this is often taken to be the surface water in the wetland in the case of a free water surface (FWS) wetland or the water in the media for a subsurface flow (SSF) wetland. A precise definition is needed to compute the change in storage. The mass balance is termed *global* when

the entire wetland water body is chosen as the system. In later chapters, it will be useful to compute the *internal* mass balance, which is based on an internal element or subdivision of the water body.

2. The *time period* for totaling the inputs and outputs must be specified. It may be desirable to express inflows and outflows in terms of rates, but these must then be averaged over the time period chosen.
3. All *inputs* and *outputs* to the chosen system must be included. The concept of mass conservation may be invoked to calculate one or a group of material flows. A partial listing of some of the inflows and outflows does not constitute a mass balance.
4. Compounds undergo chemical reactions within a wetland ecosystem. Any *production* or *destruction reactions* that occur within the boundaries of the chosen system are to be included in the mass balance. Reactions outside the boundary are not counted, because an outflow must occur to transport the chemical to the external reaction site, and that is accounted as an outflow.
5. Waterborne chemical flows are determined by separate measurements of water flows and concentrations within those waters. Therefore, an accurate *water mass balance is a prerequisite* to an accurate chemical mass balance.
6. If at all possible, it is desirable to *demonstrate closure* of the mass balance. This is achieved by independently measuring every component of the mass balance. The degree of closure is often expressed as a percentage of total inflows. Unfortunately, closure has rarely been demonstrated for any chemical in any wetland.

The foundation for chemical mass balances is the wetland water mass balance (see Chapter 2). Transfers of water to and from the wetland follow the same pattern for both surface and subsurface flow wetlands. In treatment wetlands, wastewater additions are normally the dominant flow; but under some circumstances, other transfers of water are also important. The dynamic overall water budget for a wetland is:

$$Q_i - Q_o + Q_c - Q_b - Q_{gw} + Q_{sm} + A(P - ET) = \frac{dV}{dt} \qquad (2.13)$$

where

A = wetland top surface area, m^2
ET = evapotranspiration rate, m/d
P = precipitation rate, m/d
Q_b = bank loss rate, m^3/d
Q_c = catchment runoff rate, m^3/d
Q_{gw} = infiltration to groundwater, m^3/d
Q_i = input wastewater flowrate, m^3/d
Q_o = output wastewater flowrate, m^3/d
Q_{sm} = snowmelt rate, m^3/d
t = time, d
V = water storage in wetland, m^3

It is difficult to establish detailed chemical mass balances over the wetland surface water because of the number and complexity of the possible transfers to and from the water, and their nonsteady character. It is common practice to measure only the principal inflows and outflows, and to ascribe the difference to "removal," which may be positive or negative. This lumping of all transfers to and from the water body is often unavoidable due to economic constraints. It is possible to write a general mass balance equation for a generic chemical species:

$$Q_i C_i - Q_o C_o + Q_c C_c \pm Q_{gw} C_{gw} + A(PC_p - \tilde{J}) = \frac{d(VC_s)}{dt} \qquad (6.13)$$

where

C_c = concentration in catchment runoff, g/m^3
C_{gw} = concentration in groundwater recharge or discharge, g/m^3
C_i = concentration in inflow, g/m^3
C_o = concentration in outflow, g/m^3
C_p = concentration in precipitation, g/m^3
C_s = concentration in wetland surface water, g/m^3
$\tilde{J}$ = spatially averaged removal rate, $g/m^2 \cdot d$

In Equation 6.13, bank losses and snowmelt have been omitted for the sake of simplicity. All the transfers have been lumped into one removal rate. Flow rates are instantaneous. The removal rate is the average over the entire wetland area, and the system concentration is averaged over the entire water volume.

The time period for the global mass balance is of critical importance because of the time scale of interior phenomena. Many wetlands, whether treatment or pristine natural, have long nominal detention times, which usually reflect long actual detention times. A two-week detention is not uncommon. If the wetland were in plug flow, an entering cohort of water would exit two weeks later. Clearly, same-day samples taken from inlet and outlet should not be used to compute "removals." In fact, wetland flow patterns are more complex than plug flow; the entering cohort breaks up, and pieces depart at various times after entry, some earlier and some later than the implied two-week detention. This difficulty of synchronous sampling may be alleviated in the mass balancing process by selecting a mass balance period that spans several detention times.

The removal term is the result of transfers to and from the soils and biomass compartments in the wetland, as well as of transfers to and from the atmosphere, and chemical conversions. Those biomass and soils compartments dominate the overall wetland storage and transformations for most chemicals. Therefore, the water body mass balance is very sensitive to small changes in transfers, reactions, and storages in biomass and soils. The removal rate depends very strongly on events in these solids compartments, and hence is determined in major part by the changing ecological state of the wetland. Because wetland biological processes are more or less repetitive on an annual cycle, the long-term performance of the wetland is best characterized by global mass balances that span an integer number of years. Seasonal effects require a time period of three months, which is usually long enough to avoid storage errors and detention time offset.

Removal in Equation 6.14 is an areal average. However, in most flow through wetlands, there is a strong gradient in the unaveraged removal in the direction of flow. As the downstream wetland system "boundary" is moved successively further from the inlet, the areal average removal rate decreases. The average removal rate depends on the size of the portion of the overall wetland that is chosen for the global mass balance. This weakness of the global mass balance can be corrected by using the internal mass balance that reflects distance effects.

6.4 PROCESSES THAT CONTRIBUTE TO POLLUTANT REMOVALS

A large number of wetland processes may contribute to the removal or reduction of any given pollutant. Here, some of the most important are described and the commonly used rules for quantification are presented. More details are presented in the following chapters for the most common chemicals of interest. The discussion here relates to localized phenomena. Removal processes must also be quantitatively placed in the context of internal wetland hydraulics as well as the topography and vegetative structure of the wetland.

MICROBIALLY MEDIATED PROCESSES

Many wetland reactions are microbially mediated, which means that they are the result of the activity of bacteria or other microorganisms. Very few such organisms are found free-floating; rather, the great majority are attached to solid surfaces. Often, the numbers are sufficient to form relatively thick coatings on immersed surfaces.

Transfer of a chemical from water to immersed solid surfaces is the first step in the overall microbial removal mechanism. Those surfaces contain the biofilms responsible for microbial processing, as well as the binding sites for sorption processes. The following discussion analyzes the transport of dissolved constituents to reaction sites located in the biofilms that coat all wetland surfaces. Mass transfer takes place both in the biofilm and in the bulk water phase. Roots are the locus for nutrient and chemical uptake by the macrophytes, and these are accessed by diffusion and transpiration flows. The sediment–water interface is but one such active surface; the litter and stems within the water column comprise the dominant wetted area in FWS wetlands, and the media surface is the dominant area in SSF wetlands.

Dissolved materials must move from the bulk of the water to the vicinity of the solid surface, then diffuse through a stagnant water layer to the surface, and penetrate the biofilm while undergoing chemical transformation (Figure 6.10). This sequence of events has been described and modeled in the text of Bailey and Ollis (1986), and is outlined here. The case of zero wetland background concentration will be described here; but extension to the case of nonzero background is possible.

The rate of transfer across the two films is:

$$J_{mt} = \frac{D_w}{\delta_w}(C - C_{interface}) \tag{6.14}$$

$$J_{mt} = E\, k_b\, \delta_b\, C_{interface} \tag{6.15}$$

where

C = concentration in the bulk water, mg/L = g/m^3
$C_{interface}$ = concentration at the biofilm surface, mg/L = g/m^3
D_w = diffusion coefficient in water, m^2/d
D_b = diffusion coefficient in biofilm, m^2/d
δ_b = thickness of the biofilm, m
δ_w = thickness of the stagnant boundary layer, m
E = tanh(ϕ)/ϕ, biofilm effectiveness factor, dimensionless
J_{mt} = mass transfer rate, g/m^2·d
k_b = reaction rate constant inside biofilm, d^{-1}

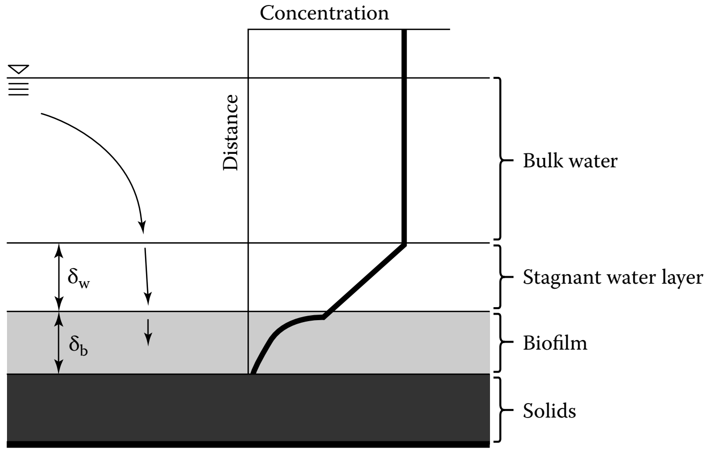


FIGURE 6.10 Pathway for movement of a pollutant from the water across a diffusion layer and into a reactive biofilm. The solid may be sediment, a litter fragment, or a submerged portion of a live plant. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

and where

$$\phi = \delta_b \sqrt{\frac{k_b}{D_b}} \tag{6.16}$$

Combining equations (the rate of transport of the pollutant from the bulk water to the biofilm) is then:

$$J_{mt} = \left[\frac{E\,k_b\,\delta_b}{1+M}\right] C = k_i C \tag{6.17}$$

where

k_i = intrinsic first order areal reaction rate constant, m/d.

$$M = \frac{E k_b \delta_b \delta_w}{D_w} \tag{6.18}$$

It is seen that this theory produces a local first-order rate of overall reaction, which depends upon biofilm properties and diffusion coefficients.

In a field situation, it is also necessary to know the area of biofilms that occupy a given area of wetland (Figure 6.11). The overall removal rate from a wetland area A_w occurs from a biofilm area of A_b, and hence the rate of removal is

$$\text{(a)} \quad J\,A_w = k_i\,A_b\,C$$

$$\text{(b)} \quad J = k_i \left(\frac{A_b}{A_w}\right) C = k_i\,a_s\,C = k\,C \tag{6.19}$$

where

a_s = biofilm area per unit wetland area, m^2/m^2

A_w = wetland area, m^2

A_b = biofilm area, m^2

k = first-order areal reaction rate constant, m/d

Data have been obtained for only a few FWS wetland systems, giving only a rough estimate of the magnitude of a_s. If there is no vegetation, and only the wetland bottom serves as the potential location of biofilms, the value of $a_s \leq 1.00$. If the emergent vegetation is considered an additional biofilm area, a dense stand of plants can yield $a_s \approx 5$. Inclusion of the litter can further increase the value to $a_s \approx 10$.

This theory has been calibrated for treatment wetlands by Polprasert and coworkers (Polprasert *et al.*, 1998; Khatiwada and Polprasert, 1999a), who determined a_s in the range 2.2–2.9. Measurements of immersed vegetation surface area were made at Arcata, California, and Houghton Lake, Michigan, and produced a_s in the range 1.0–9.0 (see Chapter 3).

Microbially mediated reactions are affected by temperature. Response is typically much greater to changes at the lower end of the temperature scale (<15°C) than the warmer range (20–35°C) (Kadlec and Reddy, 2001). Processes regulating organic matter decomposition are affected by temperature. Similarly, all nitrogen cycling reactions (mineralization, nitrification, and denitrification) are affected by temperature. The temperature coefficient (θ) varies from 1.05–1.37 for carbon and nitrogen cycling processes under isolated conditions. Phosphorus sorption reactions are least affected by temperature with θ-values of 1.03–1.12. However, treatment wetlands

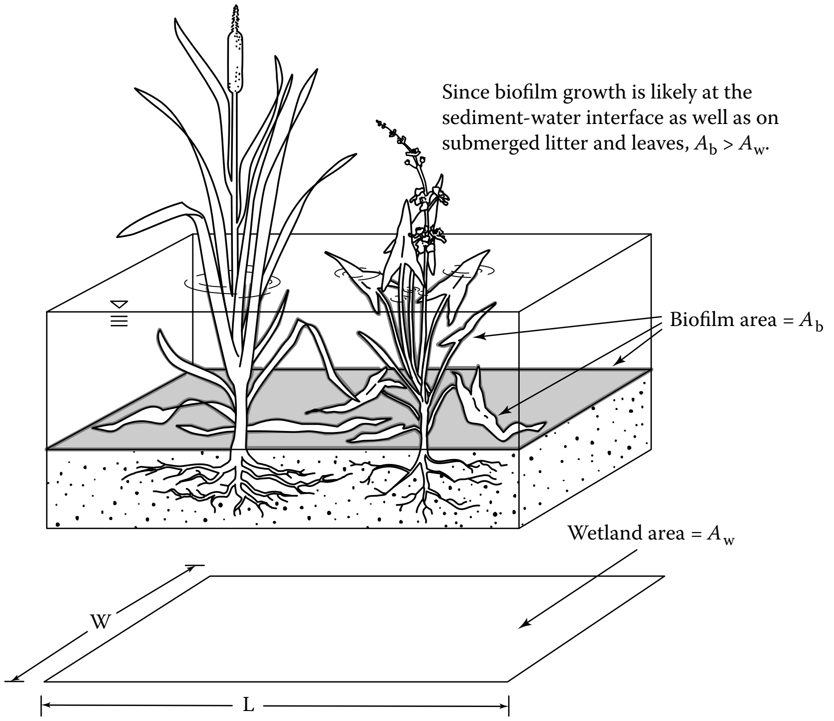


FIGURE 6.11 Biofilms dominate the sediment–water interface, as well as the surfaces of the litter and standing dead material. (Adapted from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

display lesser temperature effects because of their complexity (Kadlec and Reddy, 2001).

Chemical Networks

Several wetland chemical removal processes involve more than one reaction and more than one chemical species. Many removal reactions create products that are themselves contaminants of interest. An important example is the (microbial) sequential conversion network for nitrogenous species:

$$\text{organic N} \rightarrow \text{ammonia N} \rightarrow \text{oxidized N} \rightarrow \text{gaseous } N_2 \tag{6.20}$$

Each of the first three species is important in its own right as a potential contaminant. Both consumption and production can occur; it is, therefore, misleading to isolate one species and compute its "removal."

Another example is the reductive dechlorination of a chlorinated organic compound. Trichloroethylene has daughter products that are sequentially formed in a wetland environment:

$$\text{trichloroethylene} \rightarrow \text{dichloroethylene} \rightarrow \text{vinyl chloride} \rightarrow CO_2 + H_2O + Cl^- \tag{6.21}$$

In such cases, it is essential to utilize reaction models that account for both production and destruction. Each step may individually have a simple model, such as first order; but in combination, removal is quantitatively more complex.

Volatilization

Various processes in wetland create product gases that are released from the wetland environment to the atmosphere, such as ammonia, hydrogen sulfide, dinitrogen, nitrous oxide, and methane. Wetlands also take in atmospheric carbon dioxide for photosynthesis and expel it from respiratory processes. The mechanism of volatilization is further discussed in Chapter 5 (for nitrous oxide, methane, and carbon dioxide), Chapter 9 (for ammonia), and Chapter 11 (for hydrogen sulfide).

Sedimentation

Chapter 7 deals with the removal of suspended solids. Here, it is noted that a first-order areal removal model is the outcome of theory and practice. Many pollutants partition to suspended solids, and thus removal of those sorbed substances also is expected to follow that model:

$$J = k_{TSS} \cdot C_{TSS} \cdot K_p \cdot C \tag{6.22}$$

where

C_{TSS} = suspended matter concentration, mg/L = g/m^3
k_{TSS} = TSS removal rate constant, m/d
K_p = partition coefficient, m^3/g

The necessary connection to solids removal is the quantification of sorption.

Sorption

Partition coefficients relate the amount of sorbed pollutant to the concentration in the water under equilibrium conditions. Three types of sorption isotherms are in common use in wetland technology:

$$\begin{aligned} &\text{(a) Linear:} && C_s = K_p C \\ &\text{(b) Fruendlich:} && C_s = K_f C^n \\ &\text{(c) Langmuir:} && C_s = K_L\left(\frac{C}{C+b}\right) \end{aligned} \tag{6.23}$$

The sorption potential for the principal contaminants of interest is discussed in the chapters pertaining to those contaminants. Here, a few generalities are noted:

- Sorption is important for phosphorus during the start-up period for a treatment wetland. If initially absent in the sediments, phosphorus will be stored until the existing soils and sediments reach equilibrium with the overlying water. If initially present, phosphorus may be released.
- Sorption is important for ammonia nitrogen in intermittently dosed or operated wetlands. Short-term storage may be oxidized during drawdown periods.
- Sorption is important for hydrophilic organic chemicals, which partition strongly to the carbonaceous content of wetland sediments.
- The water-phase concentration that is experienced by wetland sediments and soils is pore water, which can have very different concentrations than the bulk water overlying those sediments and soils.
- Sorption sites are a partially renewable resource, because they may be added from the accumulation of newly formed sediments.
- Sorption may be partially irreversible, due to mineralization of sorbed materials, or to the formation of very strong chemical bonds.
- Linear sorption (Equation 6.23(a)) results in a theoretical first-order removal process at the local level.

Photodegradation

Sunlight can degrade or convert many waterborne substances. Many microorganisms, including pathogenic bacteria and viruses, can be killed by ultraviolet radiation. The effectiveness is presumptively determined by the radiation dose rate as well as the concentration of organisms. Although this is theoretically a second-order process, the sunlight dose in the wetland is relatively constant in the long run, and the elimination rate is therefore pseudo first order in the organism concentration. A wide variety of chemicals are also susceptible to removal, in one or both of two ways. Direct photolysis involves the breakdown of the molecule, usually by the ultraviolet component of the sunlight. The nitrotoluenes are examples of readily photolyzable substances. Photooxidation occurs via reactions with free radicals formed by the incident radiation, such as alkylperoxy, hydroroxyl, and singlet oxygen radicals (Howard *et al.*, 1991). Photodegradation has received essentially no attention in treatment wetland research and development.

Plant Uptake

Plants take up nutrients to sustain their metabolism. They may also take up trace chemicals found in the root zone, which may then be stored, or in some cases, expelled as gases. Uptake is by the roots, which are most often located in the wetland soils, although adventitious roots may sometimes be found in the water column. Submerged plants may absorb nutrients and metals from the water column into stems and leaves.

Vertical Diffusion in Soils and Sediments

If there is no infiltration, driven either by hydraulic head or plant transpiration, to carry dissolved contaminants to sorption and reaction sites and roots located below ground, then diffusion is the dominant mechanism for vertical downward movement of pollutants. The presence of the soil matrix prevents convection currents; therefore, the diffusive process is further restricted to molecular diffusion. The model for this process is the diffusion equation:

$$J_D = -D\frac{dC_{pw}}{dz} \tag{6.24}$$

where

C_{pw} = porewater concentration, g/m^3
D = diffusion coefficient, m^2/d
J_D = vertical diffusion flux, g/m^2·d
z = vertical distance, m

The values of diffusion coefficients in pure water are of the order of $2 - 10 \times 10^{-5}$ m^2/d at 25°C (i.e., 2.9×10^{-5} for COD, and 7.6×10^{-5} for $H_2PO_4^-$). Values in the soil pore water are likely to be lower, by about a factor of 4, because of tortuosity and porosity effects.

Some idea of the importance of the diffusive process may be gained by examining the situation of mildly eutrophic surface waters overlying a fully saturated peatland. Reddy *et al.* (1991) report soluble reactive phosphorus pore water gradients as large as 3.0 gP/m^3·m in the top 20 cm of an Everglades cattail-dominated peatland. Under these circumstances, the diffusion flux predicted by Equation 6.24 is:

$$J_D = -\left(\frac{7.6 \times 10^{-5}}{4}\right)(3)(365) = 0.02 \text{ g/m}^2 \cdot \text{yr}$$

Uptake rates in that Everglades environment were independently measured, and found to be more than an order of magnitude higher than this predicted diffusive flux (Reddy *et al.*, 1991; Richardson *et al.*, 1992). Consequently, other mechanisms were operative. An important additional mechanism is the flow of water and phosphorus pulled into the root zone of emergent macrophytes to support transpiration.

Transpiration Flux

Vertical flows of water in the upper soil horizon are also driven by plant water uptake to support transpiration. In aquatic and wetland systems with fully saturated soils or free surface water, the meteorological energy budget requires the vaporization of an amount of water sufficient to balance solar radiation and convective losses. Some of this vaporization is from the water surface (evaporation); some is from the emergent plants (transpiration). Emergent plants "pump" water from the root zone to the leaves from which water evaporates through stomata, which constitutes the transpiration loss (see Figure 4.6). In a densely vegetated wetland, transpiration dominates the combined process (evapotranspiration, which is abbreviated as *ET*) (see Chapter 4). Water for transpiration must move through the soil to the roots. That movement is vertically downward from overlying waters in FWS wetland situations, but directly from the flowing water in SSF wetlands.

Thus, transpiration has the potential to move on the order of 1 m/yr of water vertically downward to the root zone in an FWS system. That water carries with it the contaminant concentrations associated with the bottom layer of overlying water, which is the litter–benthic zone of the wetland. This flow is termed the transpiration stream (*TS*), and it draws from pore water that is typically at a concentration different from that of the bulk surface water. In turn, the plant may block a portion of the dissolved pollutant, and take up a concentration less than that of pore water. These factors combine to determine the amount of plant uptake (Trapp and Matthies, 1995; Gomez and Pardue, 2002):

$$J_U = TS \times TSCF \times C_{pw} \tag{6.25}$$

where

C_{pw} = porewater concentration, g/m^3
J_U = uptake flux, g/m^2·d
TS = transpiration stream, m/d
$TSCF$ = transpiration stream concentration factor, dimensionless

In a moderately dense emergent FWS wetland, the transpiration flux is far greater than the estimated diffusion flux.

Vertical Root Profiles

Plant roots are typically located in the top 30 cm of the soil, and most are in the top 20 cm (see Figure 2.29). However, rooting depths have been reported over a wide range. For example, for *Phragmites*, Moore *et al.* (1994) reported 10 cm, while Börner *et al.* (1998) reported 150 cm. U.S. EPA (2000a) recommends rooting media depths for FWS constructed wetlands in the range 15–40 cm. For other species, rooting depth in FWS wetlands is typically 20–30 cm. For instance, Murkin *et al.* (2000) report that roots were found entirely within the top 20 cm for *Phragmites, Typha* spp., and *Scirpus* spp. in a natural prairie marsh. Similarly, Wentz (1976) reported decreasing root biomass down to 45 cm for *Carex* spp. in the Houghton Lake wetland. Given the vertical profile of root density, there is presumptively a corresponding vertical profile in the uptake of water and chemicals by the plant. However, such differential uptake is very difficult to measure; consequently, plant uptake is usually assigned to the vertically integrated root zone.

Seasonal Cycles

Nutrient removal displays considerable seasonality for ammonia at low loadings. Accordingly, temperature is not always an acceptable surrogate for seasonality for nitrogen removal. Vegetative uptake in temperate climates is maximum during spring, at moderate temperatures, but release via decomposition is maximum during fall, also at moderate temperatures. Plants utilize phosphorus, nitrate, and ammonium, and decomposition processes release nitrogen and phosphorus back to the water. On an instantaneous basis, plant uptake can be important for many wetland systems (Kadlec, 2005d).

Accretion

One of the least studied aspects of pollutant transfer in wetlands is in the creation of new soils and sediments, with their attendant chemical content. Not all the dead plant material undergoes decomposition. Some small portions of both aboveground and belowground necromass resist decay, and form stable new accretions. Such new stores of chemicals are presumed to be resistant to decomposition. The origins of new sediments may be from remnant macrophyte stem and leaf debris, remnants of dead roots and rhizomes, and from undecomposable fractions of dead microflora and microfauna (algae, fungi, invertebrates, bacteria).

The amount of such accretion has been quantified in only a few instances for FWS wetlands (Reddy *et al.*, 1991; Craft and Richardson, 1993; Rybczyk *et al.*, 2002), although anecdotal reports also exist (Kadlec, 1997a). Quantitative studies have relied upon either atmospheric deposition markers (radioactive cesium or radioactive lead), or introduced horizon markers, such as feldspar or plaster. Either technique requires several years of continued deposition for accuracy.

6.5 CHARACTERIZATION OF INTERNAL HYDRAULICS

The removal of pollutants within a constructed wetland occurs through the diverse range of interactions between the sediments, substrate, microorganisms, litter, plants, the atmosphere, and the wastewater as it moves through the system. The dynamics of water movement through the wetland has a significant influence on the efficiency and extent of these interactions. Many of the important biogeochemical reactions rely on contact time between wastewater constituents and microorganisms and the associated substrate, whereas wastewater velocity can be an important determining factor for other pollutant removal processes, such as mass transfer. Any short-circuiting or dead zones that occur within a wetland will, consequently, have an effect on contact time as well as flow velocities and, therefore, impact on treatment efficiency. Nonideal flow patterns can have very large effects upon the removal of pollutants in wetland treatment systems (Kadlec *et al.*, 1993; Carleton, 2002). It is, therefore, necessary to consider flow pattern effects and the related mixing in the design of wetland treatment systems.

Three types of hydraulic inefficiencies may occur in treatment wetlands:

1. Internal islands and other topographical features
2. Preferential flow channels at a large distance scale
3. Mixing effects, such as water delays in litter layers and transverse mixing

The first mechanism is characterized by a gross areal efficiency, which relates to the volumetric efficiency (e_V) of the wetland, as discussed in Chapter 2. The second and third types are characterized by an equivalent set of well-mixed units in series, or other "mixing" model. All three influence a wetland's ability to improve water quality.

The main method by which wetland scientists and engineers have gained information about internal hydraulic processes is through the use of inert tracers, which provide a means of tracking the movement of water through a wetland. The theory and practice behind hydraulic investigations have predominantly evolved out of the field of chemical reaction engineering (Fogler, 1992; Levenspiel, 1995). The details of tracer testing are covered in Appendix B. Here, a brief summary is presented.

Tracer Tests

A tracer test is conducted by introducing an impulse of an inert substance into the wetland inlet at time zero. If water moved through the wetland in lock-step, such a tracer impulse would also exit as an impulse (a sharp spike of concentration). This result has never been observed in a wetland tracer test; the exit tracer is always a blurred, skewed bell-shaped curve. In the FWS wetland environment, there are mixing processes on a number of different distance scales. Expanses of open water permit development of surface wind-driven currents, which are matched by return flows in lower water layers. Deeper parallel zones in the FWS wetland carry more flow because of the depth effect on hydraulic resistance. These preferential channels may also be due to a lower vegetation density along some flow paths. A tracer impulse added to

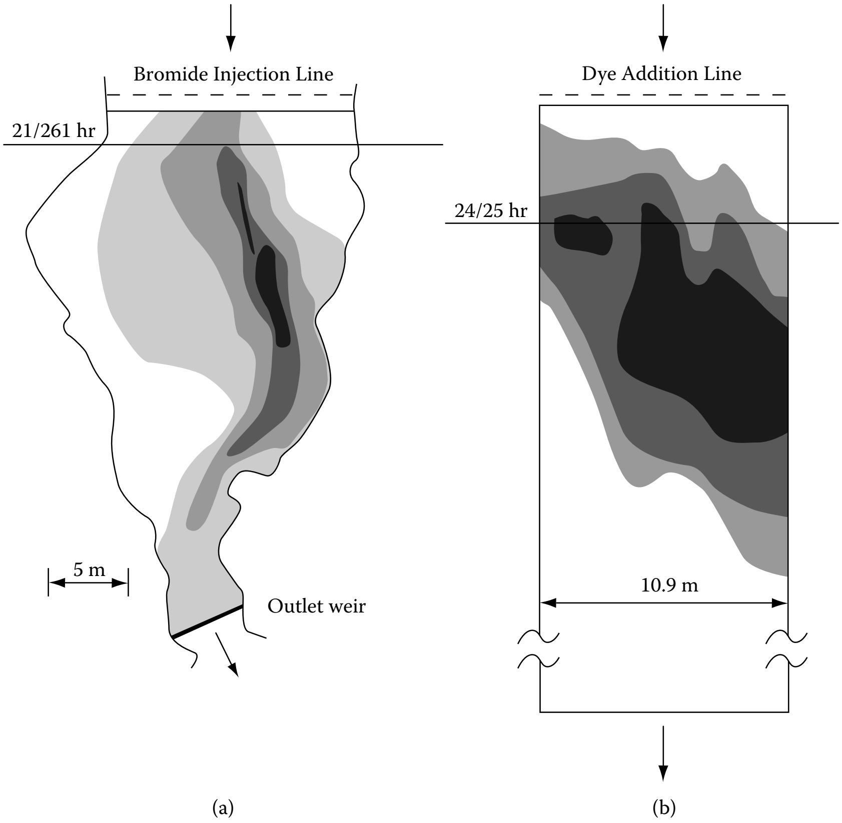


FIGURE 6.12 Tracer isopleths in a natural (a) and constructed (b) wetland. In both cases, tracer was added uniformly across the inlet width. The theoretical location of the pulse centroid is shown by the horizontal line, labeled with elapsed time and theoretical detention time. (a) *Typha orientalis* natural wetland in New Zealand. (Data from A.B. Cooper (1992) *Coupling wetland treatment to land treatment: An innovative method for nitrogen stripping.* Proceedings of the 3rd International Conference on Wetland Systems for Water Pollution Control, Australian Water and Wastewater Association and IWA, Sydney, pp. 37.1–37.9.) (b) *Typha latifolia* FWS constructed wetland in Ontario. (Data from Herskowitz (1986) *Listowel Artificial Marsh Project Report.* Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario.) (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

the incoming water provides a way to find such preferential paths as the tracer will later be found preferentially in those wetland zones. Both natural and constructed FWS wetlands display such flow variability (Figure 6.12). In particular, the results for constructed wetlands indicate that it is not possible to avoid such flow irregularities even with extreme care in construction.

There are also mixing effects in the vertical direction in FWS wetlands. Water may be moving more slowly near the bottom because of the increased drag of the dense litter layer. Those slow-moving zones exchange chemical constituents with adjacent faster-moving layers, and thus create vertical mixing. Dense plant clumps can effectively block flow even though these are of very high void fraction. Water in these clumps can exchange constituents with the adjacent microchannels by diffusive processes. All these effects combine to form a complicated overall mixing pattern. The result of such mixing is evidenced in the blurring of a tracer impulse added to the incoming water (Figure 6.12).

In a subsurface flow wetland, large-scale eddies and wind mixing are absent. However, preferential flow channels can occur on a large scale. Lateral inhomogeneities may contribute to nonuniform flow distribution across the width of the wetland (Marsteiner, 1997). Evidence of this was found for a HSSF wetland at Benton, Kentucky, by internal sampling of tracer responses (Figure 6.14). An impulse of tracer (Rhodamine WT) was added to the inlet flow to this HSSF wetland. Water was distributed across the entire width of the rectangular wetland. The observed responses were considerably different at equidistant sampling points, indicating subsurface preferential paths. Further, there is abundant evidence that vertical stratification occurs in gravel beds, with larger flows occurring at lower elevations (Fisher, 1990; Marsteiner, 1997; Drizo *et al.*, 2000). The tracer concentrations that reach the HSSF wetland effluent are there blended to form an average outlet concentration. The response of a typical HSSF wetland to an impulse tracer input is a time-delayed bell-shaped curve (Figure 6.15).

Interpretation of Data

The results of an impulse tracer test provide the volumetric efficiency (e_V) of the wetland, together with information on the distribution of detention times in the system. The first requirement of a valid tracer test is that the tracer be recovered nearly in its entirety at the wetland outlet. To that end, a simple check is made by summing the tracer at the

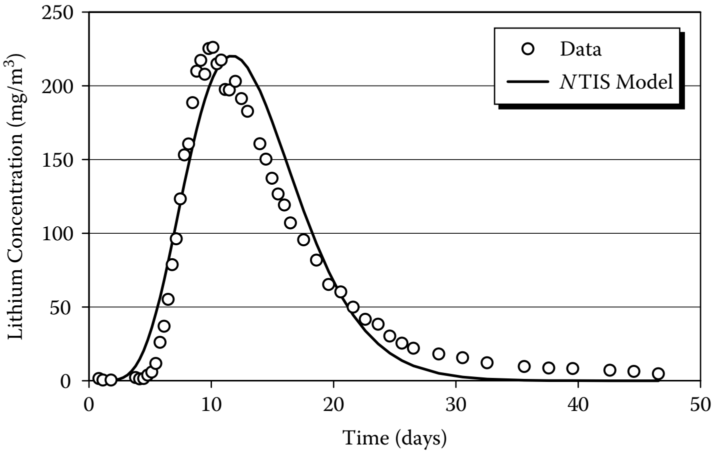


FIGURE 6.13 Results of a lithium tracer test of a 91-ha FWS wetland receiving 193,000 m^3/d. Approximately 500 kg of lithium were added. The TIS model is calibrated by about 8 TIS. (Data from Dierberg and DeBusk (2005) *Wetlands* 25(1): 8–25.)

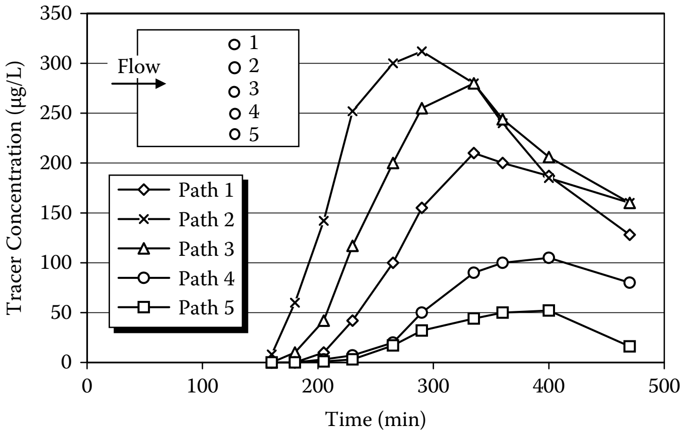


FIGURE 6.14 Tracer concentrations at five stations normal for the flow direction in Gravel Bed Wetland #3 at Benton, Kentucky. Although these traces are not complete, it is clear that more tracer arrives sooner at Station 2 than at other stations. (Data from TVA unpublished data.) (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

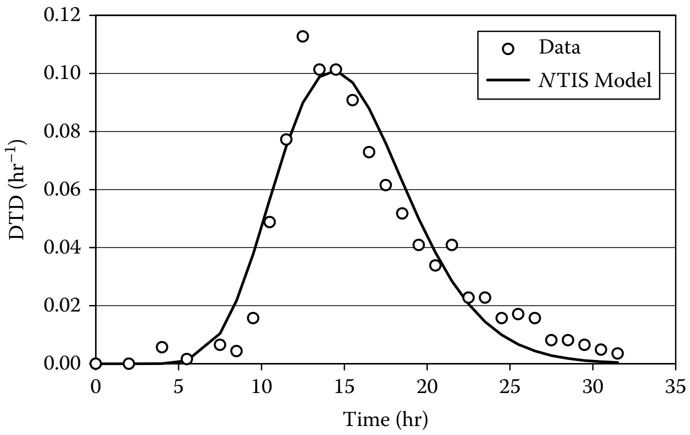


FIGURE 6.15 Response of Cell 1 at Minoa, New York, to a tracer impulse. The TIS model is calibrated to 14 TIS, and the volumetric efficiency is 75%. (Data from Marsteiner (1997) *Subsurface Flow Constructed Wetland Hydraulics.* M.S. Thesis, Clarkson University (Potsdam, New York) 130 pp.)

wetland outfall:

$$M_o = \int_0^{\infty} Q_o C\, dt = ? = M_i \tag{6.26}$$

where
C = outlet tracer concentration, mg/L
M_i = mass of tracer in, g
M_o = mass of tracer out, g
Q_o = outflow, m^3/d
t = time, days

The detention time distribution (DTD) represents the time various fractions of fluid (water in the case of the wetland) spend in the reactor, and hence is the contact time distribution for the system. In a broader context, the DTD is the probability density function for residence times in the wetland. This time function is defined by

$f(t)\Delta t =$ fraction of the incoming water which stays in the wetland for a length of time between t and $t + \Delta t$ (6.27)

where
f = DTD function, d^{-1}

For an impulse input of tracer into a steadily flowing system, the function $f(t)$ is

$$f(t) = \frac{Q_o C}{\int_0^{\infty} Q_o C\, dt} = \frac{C}{\int_0^{\infty} C\, dt} \tag{6.28}$$

The first numerator is the mass flow of tracer in the wetland effluent at any time, t, after the time of the impulse addition. The first denominator is the sum of all the tracer collected and thus should equal the total mass of tracer injected.

The mean *tracer detention time* (τ) is presumed to be the actual mean detention time, and is calculated from

$$\tau = \frac{1}{M_i} \int_0^{\infty} tQC\, dt \tag{6.29}$$

where
τ = tracer detention time, days

A wetland may have internal excluded zones that do not interact with flow, such as the volume occupied by plant materials. In a steady-state system without excluded zones, the tracer detention time (τ) equals the nominal residence time (τ_n). This is true whether the flow patterns are ideal (plug flow or well mixed) or nonideal (intermediate degree of mixing). An adsorbing tracer will produce an artificially short detention time, which may then be erroneously presumed to result from a large excluded zone. An incorrect topography may be due to either positive or negative differences between τ and τ_n. The ratio of tracer to nominal detention time is the volumetric efficiency:

$$e_V = \frac{\tau}{\tau_n} \tag{6.30}$$

where
$\tau_n = V/Q$ = nominal detention time, days
V = wetland water volume, m^3

There are a variety of reasons why the value of e_V is different from unity, as discussed previously in this chapter and in Chapter 2.

A second parameter which can be determined directly from the residence time distribution is the variance (σ^2), which characterizes the spread of the tracer response curve about the mean of the distribution, which is τ:

$$\int_0^{\infty} (t - \tau^2) f(t)\, dt = \sigma^2 \tag{6.31}$$

where
σ^2 = DTD variance, d^2

The variance of the DTD is created by mixing of water during passage or, equivalently, by a distribution of velocities of passage. This can be lateral, longitudinal, or vertical mixing, or parallel flows of different velocities. An adsorbing tracer will lead to a narrowed response pulse, and hence to an erroneously low degree of mixing. This measure of dispersive processes may be rendered dimensionless by dividing by the square of the tracer detention time:

$$\sigma_\theta^2 = \frac{\sigma^2}{\tau^2} \tag{6.32}$$

The new parameter is σ_θ^2, the *dimensionless variance* of the tracer pulse.

Models for Internal Hydraulics

Tracer testing is not an end in itself; rather, it is conducted to support the modeling and calculation of contaminant removals in the wetland system. Accordingly, the tracer information is combined with the local, or intrinsic, removal rate to produce the wetland outlet concentration. There are many candidate models that may be used, which typically involve series and parallel combinations of two idealized flow elements: perfectly mixed units and plug flow sections (Figure 6.16). It is clear from numerous studies that treatment wetlands are neither plug flow nor well-mixed. The tanks-in-series (TIS) model captures the important features of wetland DTDs that produce the skewed bell-shaped response. The TIS model requires two parameters: the number of "tanks" (N), and the mean tracer detention time (τ). As the model networks increase in complexity, such as the parallel path and finite stage models, they are able to resolve the

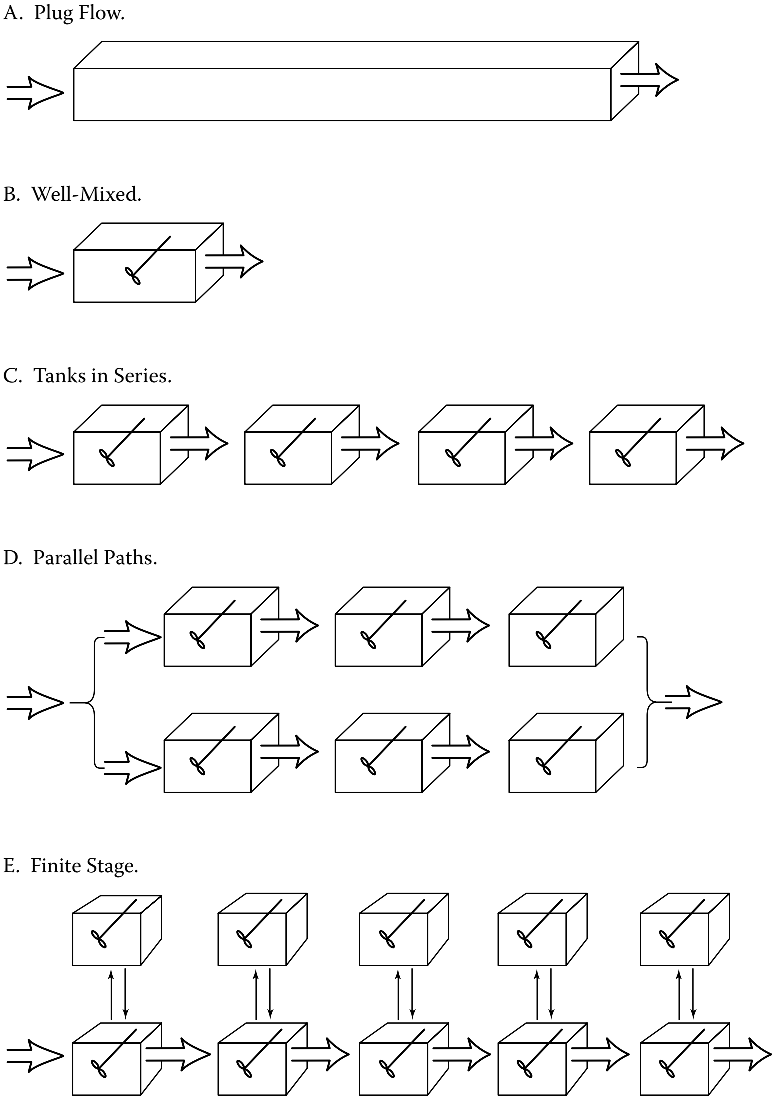


FIGURE 6.16 A sample of various models to represent wetland tracer responses. The plug flow model (A) produces an impulse output at one detention time. The well-mixed model (B) produces an exponential decline. Models (C), (D), and (E) produce skewed bell-shaped responses.

last bits of detail in the responses, but do so at the expense of adding more calibration parameters. In the extreme, it is possible to use complicated computer codes to model wetland tracer responses (Martinez and Wise, 2003a; Keefe *et al.*, 2004). In this work, the TIS model is utilized as a spreadsheet compromise between too many parameters and too little detail.

Extreme Models

The two extremes of models are the single stirred tank and plug flow. Much of the literature about flow through lakes assumes that the lake behaves as a single well-mixed unit (one tank). In contrast, rivers are often conceptualized as plug flow systems, possibly with some dispersion. Much of the early treatment wetland literature presumed plug flow, for unspecified reasons (U.S. EPA, 1988b; Water Environment Federation, 1990). The wetland tracer studies of the early 1990s made it apparent that neither extreme applied to FWS wetlands, and in many instances did not apply to HSSF systems either. Kadlec and Knight (1996) knew that plug flow did not apply, but reasoned that the plug flow assumption would be "conservative," provided that a background concentration was acknowledged. It is now known that the plug flow assumption is not always conservative (Kadlec, 1999a).

The danger in the plug flow model results from its propensity to forecast extremely low effluent concentrations, when in reality, even minor amounts of short-circuiting preclude that from happening. Therefore, the probability of design mistakes at long detention times is very high. The temptation to calculate plug flow rate constants is huge: just put numbers into Equation 6.33:

$$k = q \cdot \ln\left(\frac{C_i}{C_o}\right) \tag{6.33}$$

Other models generally require curve fitting, and are therefore more time consuming. There are two major difficulties with such calculations: (1) Equation 6.33 does not apply to synchronous samples, because of transport delay; and (2) there is no indication of the amount of variability removed by this model. If no variability is removed, the model is a useless forecaster.

The plug flow model is often an acceptable interpolator on existing data sets (Kadlec, 1999a). Thus, if high flow and low flow performance for a given system are known, a plug flow interpolation is reasonable. The difficulties arise when the model is used for extrapolation to low outlet concentrations or for extrapolation from one configuration to another. In both cases, discrepancies of a factor of two to five may easily be encountered (Figure 6.17). The parameter P in Figure 6.17 is a modification of the number of tanks in series, N, as discussed later in this chapter.

Despite these shortcomings, the wetland literature continues to espouse the plug flow formulation (e.g., Water Environment Federation, 2001; Rousseau *et al.*, 2004; Crites *et al.*, 2006). In this book, models that include the hydraulics and configuration are used. There is no loss of the ability to include near-plug-flow in those situations where it is warranted.

Tanks in Series

The TIS model is a gamma distribution of detention times:

$$g(t) = \frac{N}{\tau\,\Gamma(N)}\left(\frac{Nt}{\tau}\right)^{N-1} \exp\left(-\frac{Nt}{\tau}\right) \qquad (6.34)$$

where

$\Gamma(N)$ = gamma function of N, = $(N-1)!$, factorial, if N is an integer, d^{-1}
N = number of tanks (shape parameter), unitless
t = detention time, d
τ = mean detention time, d

When $N = 1$, the gamma distribution becomes the exponential distribution. Both the gamma distribution and the gamma function are readily available in handbooks (e.g., Dwight, 1961), or as computer spreadsheet tools (e.g., GAMMADIST and GAMMALN in Excel™). Equation 6.34 may easily be fit to tracer data by selecting N and τ to minimize error (e.g., SOLVER in Excel™). This is a gradient search procedure, in which N and τ are selected to minimize the sum of the squared errors (SSQE) between the DTD model and the data. Old textbook methods involve computation of the first and second moments of the experimental outlet concentration distribution, which are related to tracer detention time and the number of TIS, respectively. A serious failing of that moment method is that minor concentration anomalies on the "tail" of the concentration response curve may yield spurious parameter values, and bad fits of the main part of the DTD. The mode of the distribution (peak time), and its height, are also useful in determining N and τ, but the peak may not be well defined. For purposes of parameterization, it is noted that for the TIS model or gamma DTD distributions:

$$\sigma_\theta^2 = \frac{1}{N} \qquad (6.35)$$

$$\frac{\tau - \tau_{peak}}{\tau} = \frac{1}{N} \qquad (6.36)$$

Examples of least squares gamma fits of tracer data are shown previously in Figures 6.13 and 6.15. It is to be noted that although gamma distributions describe TIS mixing, the converse is not true. A gamma distribution of detention times does not imply the existence of turbulent mixing. Indeed, a gamma distribution may also arise from totally unmixed, separate travel paths with different velocities (Kadlec, 2000).

In the limit as N becomes very large, the gamma distribution becomes the plug flow (PF) distribution, with all water departing after exactly one detention time. This limiting case does not exist for treatment wetlands. Reported literature values are $N = 4.1 \pm 0.4$ (mean ± SE) for FWS wetlands, and $N = 11.0 \pm 1.2$ for HSSF wetlands (Tables 6.1–6.2). However,

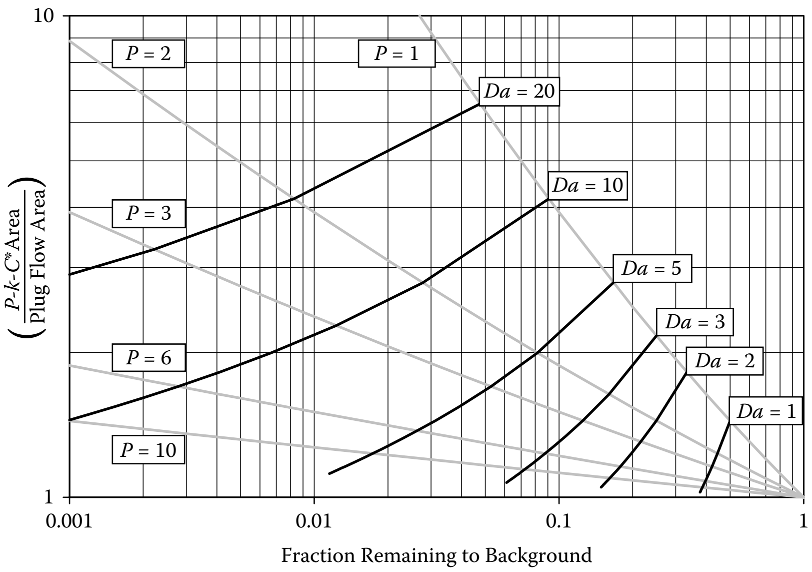


FIGURE 6.17 Comparison of plug flow and P-k-C^* areas required for specified percentage reductions. Note that the areas are much larger for low values of P, and for higher Damköhler numbers ($Da = k/q$). The fraction remaining to background is $(C - C^*)/(C_i - C^*)$.

TABLE 6.1
A Sampling of Tracer Study Results from Various Sizes of FWS Wetlands

State or Country	Project	Vegetation	Name	Size (m^2)	Depth (cm)	Tracer	Recovery (%)	nHRT (days)	Tracer HRT (days)	Volumetric Efficiency (%)	*N*TIS	Source
Florida	Everglades	SAV	Mesocosms	4	40	Lithium	75	3.6	4.0	111	4.0	1
Florida	Everglades	SAV	Mesocosms	4	80	Lithium	90	7.2	7.7	107	7.7	1
Florida	Everglades	SAV	Mesocosms	4	120	Lithium	96	10.8	9.0	83	9.0	1
Florida	Everglades	Algae	Mesocosms	6	44	Lithium	86	11.1	13.1	118	1.4	1
Florida	Everglades	Algae	Mesocosms	18	34	Lithium	75	6.5	16.6	256	1.4	1
Australia	Richmond	Open Water	Open Water	400	0.5	RWT	—	5.70	4.63	81	1.6	2
Australia	Richmond	SAV	SAV	400	0.5	RWT	—	6.30	5.00	79	5.5	2
Florida	Champion	Mixed Emergent	E	1,000	34	Lithium	82	0.97	0.88	91	10.7	3
Florida	Champion	Mixed Emergent	F	1,000	68	Lithium	105	3.38	2.50	74	2.0	3
Florida	Everglades	SAV	Test Cells	2,000	71	RWT	83	5.6	5.9	105	5.9	1
Florida	Everglades	Algae	Test Cells	2,000	48	Lithium	82	10.6	11.6	109	3.1	1
Florida	Champion	Mixed Emergent	C	2,000	34	Lithium	90	2.59	1.12	43	4.0	3
Florida	Champion	Mixed Emergent	D	2,000	57	Lithium	58	11.83	9.66	82	3.5	3
Florida	Everglades	Cattails	Test Cells	2,457	36	Lithium	95	3.6	2.1	60	4.3	1
California	Sacramento	Bulrushes	7B	2,926	55	Lithium	76	4.8	4.7	98	4.2	4
California	Sacramento	Bulrushes	9B	2,926	56	Lithium	78	4.8	6.4	133	5.2	4
Florida	Champion	Mixed Emergent	A	4,000	34	Lithium	64	7.48	4.68	63	6.3	3
Florida	Champion	Mixed Emergent	B	4,000	48	Lithium	37	21.45	17.40	81	3.6	3
Arizona	Tres Rios	Bulrushes	H2	9,100	49	Bromide	95	4.7	2.8	60	5.8	5
Arizona	Tres Rios	Bulrushes	H1	9,200	49	Bromide	80	4.3	3.3	77	8.6	5
Arizona	Tres Rios	Bulrushes	C2	12,800	52	Bromide	99	3.0	1.8	62	7.2	5
Arizona	Tres Rios	Bulrushes	C1	13,400	67	Bromide	79	3.0	2.4	79	6.1	5
Illinois	Des Plaines	Cattails	EW3	20,000	60	Lithium	98	12.3	8.4	68	2.7	6
Florida	Iron Bridge	Mixed Emergent	3	56,680	—	Bromide	91	2.57	1.34	52	1.4	7
Florida	Iron Bridge	Mixed Emergent	4	56,680	—	Bromide	95	8.97	2.28	25	2.1	7
Florida	Iron Bridge	Mixed Emergent	7	117,409	—	Bromide	97	13	2.63	20	3.1	7
Florida	Iron Bridge	Mixed Emergent	8	121,457	—	Bromide	112	2.95	1.24	42	0.3	7
Florida	Iron Bridge	Mixed Emergent	1	230,769	—	Bromide	109	4.53	1.38	30	1.3	7

State or Country	Project	Vegetation	Name	Size (m^2)	Depth (cm)	Tracer	Recovery (%)	nHRT (days)	Tracer HRT (days)	Volumetric Efficiency (%)	*N*TIS	Source
Florida	Lakeland	Cattails	Cell 4	300,000	—	Lithium	84	—	35.00	—	2.8	8
Florida	Lakeland	Cattails	Cell 2	770,000	—	Lithium	67	—	5.70	—	0.3	8
Florida	Lakeland	Mixed Emergent	Cell 1	810,000	—	Lithium	75	—	0.70	—	1.5	8
Florida	Everglades	SAV	ENR4	1,470,000	74	RWT	83	—	4.3	—	3.5	1
Florida	Everglades	SAV	ENR4	1,470,000	76	RWT	60	—	2.8	—	1.2	1
Florida	Everglades	Mixed Emergent	ENR2	4,140,000	83	Lithium	—	—	11.3	—	5.6	1
Florida	Everglades	Mixed Emergent	ENR1	5,780,000	58	Lithium	—	—	13.4	—	7.9	1
						Mean	**84**	**6.7**	**6.5**	**82**	**4.1**	
						SD	**3**			**8**	**0.4**	

Sources:

1. Unpublished data, South Florida Water Management District
2. Bavor *et al.* (1988)
3. Unpublished data, Champion Paper Co.
4. Nolte and Associates (1998b)
5. Unpublished data, City of Phoenix
6. Kadlec (1994)
7. Wang and Jawitz (2006)
8. Unpublished data, City of Lakeland

TABLE 6.2
A Sampling of Tracer Study Results from Various Sizes of HSSF Wetlands

State or Country	Project	Vegetation	Name	Size (m^2)	Depth (cm)	Tracer	Recovery (%)	nHRT (days)	Tracer HRT (days)	Volumetric Efficiency (%)	*N*TIS	Source
Quebec	Saint-Damien	*Phragmites*	T2	1.0	25	Lithium	63	4.30	5.13	119	3.4	1
Quebec	Saint-Damien	Cattails	T3	1.0	25	Lithium	65	5.95	4.54	76	2.5	1
Tennessee	Baxter	Bulrush	B1	5.9	45	Lithium	94	1.75	2.01	115	5.3	2
Tennessee	Baxter	Bulrush	B2	5.9	45	Lithium	100	1.00	0.83	83	6.0	2
Tennessee	Baxter	Bulrush	K	5.9	45	Lithium	59	4.88	6.71	138	4.9	2
Tennessee	Baxter	Bulrush	I	5.9	45	Lithium	160	1.61	1.67	103	5.6	2
Tennessee	Baxter	Bulrush	N	11.8	45	Lithium	89	1.75	1.80	103	7.2	2
California	UC–Davis	Cattails	Old	15	95	Bromide	94	9.7	8.66	89	23.4	3
California	UC–Davis	None	New	15	95	Bromide	99	9.7	11.13	115	24.1	3
Spain	Barcelona	*Phragmites*	A1	55	50	Bromide	86	5.13	5.25	102	3.4	4
Spain	Barcelona	*Phragmites*	A2	55	50	Bromide	106	5.13	5.17	101	3.4	4
Spain	Barcelona	*Phragmites*	B1	55	50	Bromide	99	5.13	4.50	88	5.3	4
Spain	Barcelona	*Phragmites*	B2	55	50	Bromide	92	5.13	7.00	137	8.3	4
Spain	Barcelona	*Phragmites*	C1	55	50	Bromide	94	5.13	5.50	107	6.7	4
Spain	Barcelona	*Phragmites*	C2	55	50	Bromide	105	5.13	6.54	128	11.1	4
North Carolina	New Hanover	Bulrush	SSF-R	61	60	Lithium	98	5.3	4.61	87	6.8	5
North Carolina	New Hanover	None	SSF-C	61	60	Lithium	96	2.9	2.58	89	7.2	5
New Zealand	Paraparaumu	—	Point	132	78	RWT	—	4.00	2.55	64	4.5	6
New Zealand	Paraparaumu	—	Manifold	132	78	RWT	—	4.00	3.60	90	5.5	6
Minnesota	Grand Lake	Cattails	Cell 1	182	60	Bromide	84	15.1	12.10	80	4.8	7
Australia	Richmond	Bulrush	Trench 3	400	69	RWT	—	4.23	4.02	95	13.8	8
Australia	Richmond	Cattails	Trench 5	400	68	RWT	—	4.20	3.08	73	21.0	8
Australia	Richmond	None	Trench 2	400	45	RWT	—	3.10	3.41	110	25.2	8
France	Curienne	*Phragmites*	Mar-00	605	72	Chloride	96	3.00	2.00	67	10.0	9
France	Curienne	*Phragmites*	May-00	605	72	Chloride	82	1.48	1.79	121	16.0	9
France	Curienne	*Phragmites*	Jul-00	605	72	Chloride	87	4.34	3.29	76	14.0	9
France	Curienne	*Phragmites*	Dec-00	605	72	Chloride	78	2.99	1.88	63	7.0	9
France	Curienne	*Phragmites*	Mar-01	605	72	Chloride	91	1.28	1.29	101	11.0	9
France	Curienne	*Phragmites*	May-01	605	72	Chloride	93	3.03	1.50	50	9.0	9
France	Curienne	*Phragmites*	Jul-01	605	72	Chloride	79	4.16	3.33	80	14.0	9
France	Curienne	*Phragmites*	Dec-01	605	72	Chloride	92	3.33	1.83	55	7.0	9

State or Country	Project	Vegetation	Name	Size (m²)	Depth (cm)	Tracer	Recovery (%)	nHRT (days)	Tracer HRT (days)	Volumetric Efficiency (%)	*N*TIS	Source
New York	Minoa	*Phragmites*	Cell 3	1,000	76	Bromide	81	0.60	0.47	78	10.6	10
New York	Minoa	*Phragmites*	Cell 1	1,700	76	Bromide	128	0.85	0.64	75	14.3	10
New York	Minoa	*Phragmites*	Cell 2 Bottom	1,750	76	Bromide	95	0.85	0.66	78	10.7	10
New York	Minoa	*Phragmites*	Cell 2 Top	1,750	76	Bromide	100	0.85	0.64	77	16.8	10
New York	Minoa	*Phragmites*	Cell 2AB	1,750	76	Bromide	116	0.83	0.73	88	21.6	10
New York	Minoa	*Phragmites*	Cell 2 Clogged	1,750	76	Bromide	45	5.43	4.50	83	34.4	10
						Mean	**92**			**91**	**11.0**	
						SD	**4**			**4**	**1.2**	

Sources:

1. Chazarenc *et al.* (2004)
2. George *et al.* (1998)
3. Grismer *et al.* (2001)
4. García *et al.* (2004b)
5. Rash and Liehr (1999)
6. Shilton and Prasad (1996)
7. Kadlec *et al.* (2003b)
8. Bavor *et al.* (1988)
9. Chazarenc *et al.* (2003)
10. Marsteiner (1997)

the range of values is quite large, and depends strongly on wetland configuration, which will be discussed in subsequent chapters.

This TIS quantification of internal hydraulics forms the basis for development of reaction models for treatment wetlands. The TIS hydraulic model is flexible enough to describe both mixing and preferential flow paths for a wide range of hydraulic efficiencies.

Plug Flow with Dispersion

Another model uses a dispersion process superimposed on a plug flow model (PFD). Mixing is presumed to follow a convective diffusion equation. A 1-D spatial model is chosen, because analytical expressions are available for computation of pollutant removal for the 1-D case (Fogler, 1992). A 2-D version requires the 2-D velocity field, which has yet to be determined for any operating treatment wetland. The tracer mass balance equation includes both spatial and temporal variability:

$$\left(D\frac{\partial^2 C}{\partial x^2} - \frac{\partial(uC)}{\partial x}\right) = \frac{\partial C}{\partial t} \tag{6.37}$$

where
u = velocity, m/d
D = dispersion constant, mm^2/d
x = distance from inlet toward outlet, m

The appropriate wetland boundary conditions for this mass balance are known as the *closed-closed* boundary conditions (Fogler, 1992). These imply that no tracer can diffuse back from the wetland into the inlet pipe, nor back up the exit structure at the wetland outlet. These are different from the *open-open* boundary conditions that are appropriate for river studies. There are analytical, close-form solutions to the latter case, which has led to their repeated misapplication to wetlands (Bavor *et al.*, 1988; Stairs, 1993). There are no closed form solutions to the wetland case, but numerical solutions to the closed-closed tracer mass balance have been available for more than three decades (Levenspiel, 1972). It is possible to calculate the dispersion constant that fits a particular data set, although there are issues of accuracy. This model is not advocated here, because the PFD model is only marginally applicable to treatment wetlands (see Appendix B).

The dimensionless parameter that characterizes Equation 6.37 is the Peclet number (Pe), or its inverse, the wetland dispersion number ($\mathfrak{D}$):

$$\mathfrak{D} = \frac{D}{uL} = \frac{1}{\text{Pe}} \tag{6.38}$$

where
Pe = Peclet number, dimensionless
$\mathfrak{D}$ = wetland dispersion number, dimensionless
L = distance from inlet to outlet, m

The two results of interest from modeling of the pulse test are the tracer detention time and the dimensionless variance:

$$\tau = \frac{L}{u} \tag{6.39}$$

$$\sigma_\theta^2 = 2\mathfrak{D} - 2\mathfrak{D}^2(1 - e^{-1/\mathfrak{D}}) \tag{6.40}$$

The principal problems with application of the PFD model to wetlands have to do with meeting the assumptions implicit in the model. Levenspiel (1972) notes:

> In trying to account for large extents of backmixing with the dispersion model we meet with numerous difficulties. With increased axial dispersion it becomes increasingly unlikely that the assumptions of the dispersion model will be satisfied by the real system.

The condition of an intermediate amount of axial dispersion is nominally taken to be $D/uL > 0.025$ (Levenspiel, 1972), which corresponds to about twenty TIS. Therefore, on average, FWS wetlands are not within the acceptable mixing range, but SSF systems may be marginally within range (see Tables 6.1–6.2). However, a bigger obstacle to accepting the PFD model consists of the concentration profiles that are predicted for reactive constituents, which will be addressed in Section 6.6.

6.6 REACTION RATE MODELS

In this section, the concepts of local pollutant reduction are blended with wetland hydraulic considerations, and environmental and ecosystems features, to develop pollutant removal models.

INTRINSIC CHEMISTRY

The removal of a contaminant may depend upon the local concentrations of that contaminant in any of a number of ways, depending upon the mechanism(s) or pathways involved. Additionally, other substances may be involved in the conversion process.

Zero Order

The most simplistic quantitative model for contaminant reduction is a constant rate of removal, termed *zero-order removal*, because it does not depend upon how much of the contaminant is present at a given location. The local load removal is given by

$$J = \text{constant} \tag{6.41}$$

where
J = removal per unit area, or load removed, $g/m^2{\cdot}d$

Such a constant rate of consumption has been postulated in only a few instances for treatment wetland situations (Seidel, 1966; Horne, 1995; Kadlec and Srinivasan, 1995).

First Order

As seen earlier in this chapter, many individual wetland processes are basically first order, such as mass transport, volatilization, sedimentation, and sorption. It is, therefore, not unreasonable to presume that these behave similarly in combination, at least over some range of pollutant concentration. The local removal rate equation is:

$$J = kC \tag{6.42}$$

where
C = concentration, g/m^3
J = removal per unit area, or load removed, g/m^2·d
k = rate coefficient, m/d

This rate equation is the most prevalent in treatment wetland literature, although in many instances, it is only presumptively advocated.

Saturation: Monod

Many biologically mediated reactions are first order only for concentrations lower than a saturation value. The premise is based upon the limited ability of the biological community to respond to increases in chemical availability, and this concept is implemented in models of other wastewater treatment technologies (Metcalf and Eddy Inc. 1991). Such a model interpolates between zero and first-order limits:

$$J = k\left(\frac{C}{K + C}\right) \tag{6.43}$$

where
K = half-saturation constant, g/m^3

For low values of $C << K$, this is a first-order model. For high values of $C >> K$, it is a zero-order model. This model has been infrequently implemented for treatment wetlands. Kadlec (1997a) reported that this model was appropriate for phosphorus removal in FWS wetlands, with a half-saturation concentration of 0.8 mg/L. Mitchell and McNevin (2001) suggested that a Monod model was appropriate for BOD removal, but did not offer any calibrations or half-saturation concentrations.

Limiting Reactants

Some removal processes require a second reactant to achieve the transformation or removal process. Nitrification requires oxygen, and denitrification requires a carbon source. In such cases, removal rates may be limited by the supply of the second reactant, in addition to the concentration of the contaminant in question. Other treatment technologies utilize removal models that incorporate such supply limitations. For instance, for microbial nitrification

$$k = k_{\max}\left(\frac{C_{DO}}{K_{DO} + C_{DO}}\right) \tag{6.44}$$

where
K_{DO} = dissolved oxygen half-saturation constant, g/m^3

The value of K_{DO} is suggested to be about 1.0 mg/L (U.S. EPA, 1993b). In a similar fashion, the carbon limitation for denitrification may be expressed as

$$k = k_{\max}\left(\frac{C_{OrgC}}{K_{OrgC} + C_{OrgC}}\right) \tag{6.45}$$

where
K_{OrgC} = organic carbon half-saturation constant, g/m^3

The value of K_{OrgC} is suggested to be about 0.1 mg/L (U.S. EPA, 1993b). As a result, the effect of the carbon supply is not large, unless that supply is very low.

In virtually all of the treatment wetland literature, supply limitations are implicit in the overall rate constants that are reported. Nonetheless, there are some calibrations available for specific situations (McBride and Tanner, 2000; Langergraber, 2001).

Return Fluxes and Background Concentrations

For many chemicals, the return rate to the water from the static compartments of the ecosystem—the soils and biomass—can be a significant (negative) contribution to the net rate. There is at present no scientific study to provide guidance on modeling this transfer. Therefore, the simplest option is used here: a constant (zero-order) return rate. The lumped rate equation for the net reduction of a chemical with no precursors is therefore written as

$$J = kC - r^* = k(C - C^*) \tag{6.46}$$

where
C^* = background concentration, g/m^3
k = removal rate constant, m/d (or with unit conversion, m/yr)
r^* = return rate of chemical, g/m^2·d

In the terminology of reaction engineering, the model is first order in the forward direction, and zero order in the reverse direction. The concentration C^* is achieved when there is no net uptake or conversion of the chemical in question, and is therefore termed the "background" concentration. When inlet waters have $C > C^*$, there will be a decrease with travel or time to this background concentration. When inlet waters have $C < C^*$, there will be an increase up to this background concentration.

There are several possible reasons for the existence of a real or apparent nonzero background concentration for a specific chemical constituent. First, there may be some portion of the incoming chemical that is resistant to storage or conversion in the wetland environment. This is particularly possible when the concentration measures a lumped set of species, one or more of which may be resistant to degradation in the wetland. For instance, total phosphorus in water may exist as particulate and dissolved, organic, and inorganic forms. Some portion of the organic phosphorus may be highly resistant to uptake by the biogeochemical cycle. An extreme example would be the organophosphate pesticide diazinon, which is not efficiently degraded in wetlands. However, more benign sources may contain a biologically unavailable fraction, by virtue of the size and character of the molecules embodying the phosphorus (Proctor *et al.*, 1999). Such phosphorus fractions may pass through the system untouched.

The second reason for a nonzero background concentration is the association of the chemical with particulates. Because the chemical is associated with (sorbed or incorporated in the structure) suspended particulate matter, a nonzero background level of TSS entails a nonzero background level of the chemical. For instance, at the Des Plaines, Illinois, wetlands, the export of 8 mg/L of TSS carried 16 μg/L of phosphorus, which is due to a phosphorus content of 0.2% in the exported TSS. Although TSS is notoriously difficult to measure inside wetlands, background levels of 5–10 mg/L are commonly found in densely vegetated systems (see Chapter 7).

The third reason is a set of wetland processes that provide inputs distributed across the entire areal extent of the system. Groundwater discharge and rainfall may bring a specific compound into all portions of a wetland (Raisin *et al.*, 1999). The chemical may be utilized in the biogeochemical cycle, which is also distributed across the entire wetland area. That same cycle can produce return of the substance to the water column, usually by the processes of decomposition and leaching (Kadlec, 1997a).

Fourth, there is seasonality. Dry seasons may be accompanied by loss of surface water, and dry-out of the surface sediments in the wetland. The organics in the surface sediments may then be oxidized, resulting in the mineralization of previously organic-bound substances. Upon subsequent rewetting, these mineralized materials may dissolve and contribute to surface water concentrations.

Another factor influencing concentration gradients, and the possibility of plateaus, is hydraulic bypass of the reactive wetland environment. Bypassed water carries with it the inlet substances, which may reblend with treated water at downstream wetland locations (Kadlec, 2000). This process cannot create a true plateau or background, but may easily lead to an inferred background concentration, derived from extrapolation of gradients in the upstream portion of the system. For some chemicals, very few treatment wetlands extend beyond the zone of total containment, and such extrapolation, via curve fitting, is the norm rather than the exception. As shown in Kadlec (2000), the nature of internal flow patterns leads to a data-fitted background concentration, which varies strongly with the hydraulic loading rate to the wetland. Higher loading rates lead to higher background concentrations for flow through wetlands. In this chapter, methods will be set forth to minimize this effect.

Batch versus Flow Systems

There is a strong but incorrect presumption often made that batch and continuous flow wetland systems are equivalent if travel time is exchanged for batch time. There are two potential reasons for major differences: internal hydraulics and ecosystem gradients.

A batch system will tend to be spatially uniform. The concepts of short-circuiting and dead zones do not apply. The components of the ecosystem, including plants, algae, and microbes, are exposed to a time-changing water chemistry, which may foster time variable consortia of microbes (Stein *et al.*, 2003). Consequently, the hydraulic model is always presumed to be a well-mixed batch. In laboratory mesocosm environments, the water mass balance often is not influenced by water losses or gains. In field situations, the hydraulic efficiency is 100%, because the entire wetland is filled with water. Full-scale batch treatment wetlands operated at Humboldt, Saskatchewan (Lakhsman, 1981), and continue to be operated by Ducks Unlimited Canada at Oak Hammock, Manitoba. For a case of no water losses or gains, the batch contaminant mass balance is

$$V\frac{dC}{dt} = JA = kA(C - C^*) \qquad (6.47)$$

where
A = wetland area, m^2

Over a given time period, this mass balance integrates to

$$\frac{(C - C^*)}{(C_i - C^*)} = \exp\left(-\frac{kt}{h}\right) \qquad (6.48)$$

where
C_i = starting concentration, g/m^3
h = wetland free water depth, m

This model has been calibrated for batch microcosms, and the ramifications of different statistical fitting procedures discussed by Stein *et al.* (2006b).

Continuous Flow Wetlands

A continuous flow system will not be spatially uniform. Plants, algae, and microbes vary in type and density along the path of water travel. The concepts of short-circuiting and dead zones do apply. The hydraulic model must account for these effects—through the use of the TIS model, for instance.

The TIS Model

Water passes through N tanks in series, and loses contaminant in each (Figure 6.18). For the case of no water losses or gains,

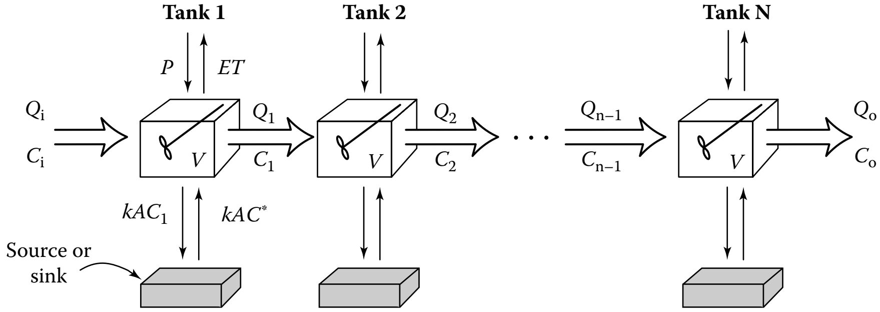


FIGURE 6.18 The tanks-in-series (TIS) model for wetland hydraulics and contaminant removal.

the steady flow contaminant mass balance for the jth tank is

$$(QC_{j-1} - QC_j) = kA(C_j - C^*) \tag{6.49}$$

where
C_j = concentration in and leaving tank j, g/m^3

For the entire sequence of tanks, these mass balances combine to

$$\frac{(C - C^*)}{(C_i - C^*)} = \left(1 + \frac{k\tau}{Nh}\right)^{-N} \tag{6.50}$$

Note that there are two reaction parameters in this model: the rate constant (k) and the hydraulic parameter (N).

In Equation 6.50, it has been presumed that the rate constant (k) does not vary with the time of exposure to the wetland. This is typically not the case for mixtures of contaminants, such as BOD, and modification is required for such situations, as will subsequently be set forth.

The PFD Model

The first-order concentration reduction produced by the PFD model is well known (see, e.g., Fogler, 1992):

$$\frac{(C - C^*)}{(C_i - C^*)} = \frac{4b \exp\left(\frac{\text{Pe}}{2}\right)}{(1+b)^2 \exp\left(\frac{b\text{Pe}}{2}\right) - (1-b)^2 \exp\left(\frac{-b\text{Pe}}{2}\right)} \tag{6.51}$$

$$\begin{aligned} &\text{(a)} \qquad b = \sqrt{1 + 4\frac{Da}{\text{Pe}}} \\ &\text{(b)} \qquad Da = \frac{k\tau}{h} \\ &\text{(c)} \qquad \text{Pe} = \frac{uL}{D} \end{aligned} \tag{6.52}$$

where
D = dispersion coefficient, m^2/d
L = wetland length, m
u = average water velocity, m/d

Note that there are also two reaction parameters in this model: the rate constant (k) and the dispersion coefficient (D).

Although the PFD model has been advocated for wetlands (e.g., Pardue *et al.*, 2000), it is doubtful that it is the most appropriate model of comparable complexity. The DTDs for FWS wetland systems are characterized by a large amount of apparent dispersion, with $0.07 \le D/uL \le 0.35$ (Kadlec, 1994a). The PFD model is not suitable under those circumstances (Levenspiel, 1995). The dispersion coefficient describes eddy transport of water elements both upstream and downstream. In FWS wetlands, such mixing may not occur because flow is often predominantly laminar.

Longitudinal Profiles

Equations 6.50 and 6.51 represent the input–output concentration behavior of different hydraulic models with first-order rate expressions. Longitudinal concentration profiles may also be derived from these models. In the case of the TIS model, the result is:

$$\frac{(C - C^*)}{(C_i - C^*)} = \left(1 + \frac{k\tau y}{Nh}\right)^{-N} \tag{6.53}$$

where
y = fractional distance through the wetland, dimensionless

This profile is a smoothly decreasing concentration that starts at the inlet concentration and levels off at a plateau value of C^*. In theory, the parameters of the model could be determined from an analysis of longitudinal transect data for C and y. In practice, there are nearly insurmountable difficulties that arise from three principal reasons. First, if there is time variability in the inlet concentration, profiles reflect that effect, together with a transport delay. In such a dynamic situation, profile data must be averaged over a sufficiently long period to determine mean behavior. Second, it is typically impossible to determine where to take a sample across the width of the wetland so that it is spatially representative. There is usually a bias toward open water areas, because of the ease of obtaining the sample. However, such open water is often a preferential flow path (short-circuit). This effect has been clearly elucidated by (Dierberg *et al.*, 2005).

It is easily seen that any given sample location may be either inside or outside of a preferential flow path (see Figure 6.12). Third, it is not enough to collect a set of spatially uniform samples in an effort to gain access to all flow paths. Spatially uniform sampling across the flow direction will produce an average that is substantially different from the mean (flow-weighted) concentration at that distance (Levenspiel and Turner, 1970; Kadlec, 1999d).

If there is a reasonably long period of averaging, the first obstacle may be overcome. The second and third obstacles may be eliminated by sampling of deep zones perpendicular to flow along the transect, but there is no guarantee. Figure 6.19a shows such deep zone sampling for ammonia, averaged over half a year at Sacramento, California. Decreasing profiles result that display a smooth decline, but there is an abrupt change at the system outlet, indicating some remaining difficulties. Figure 6.19b shows such deep-zone sampling for oxidized nitrogen, averaged over two years at Tres Rios, Arizona. On average, there is a slight increase in the inlet region

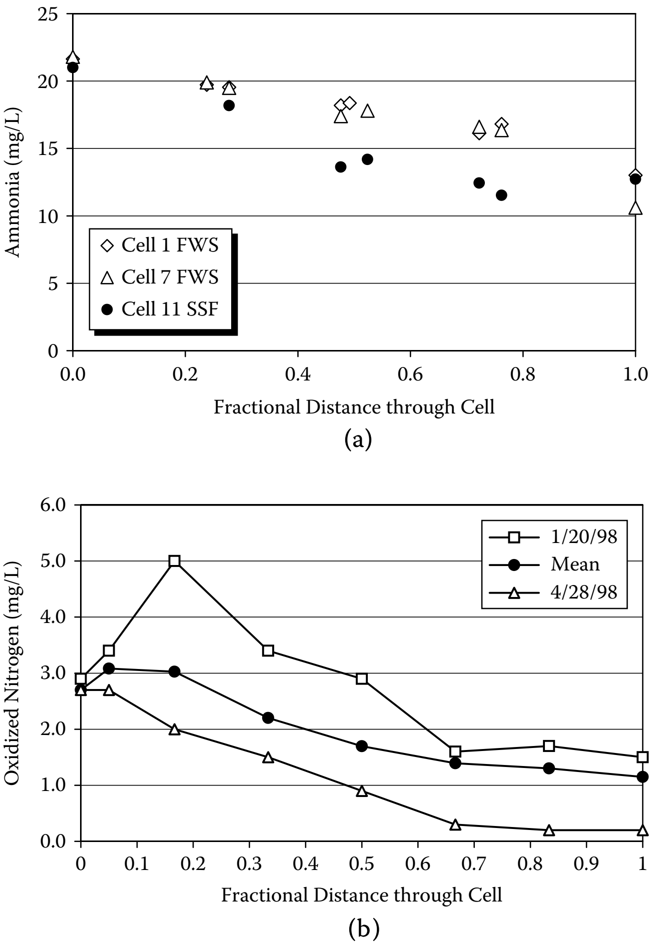


FIGURE 6.19 Examples of transect data for treatment wetlands. (a) Ammonia nitrogen for the Sacramento wetlands, spanning spring and summer 1997 (Data from Nolte and Associates, 1998a). (b) Oxidized nitrogen for the Tres Rios FWS wetland cell H1. The mean represents 32 transects over two years, and two extremes are shown. (Unpublished data from city of Phoenix.)

of the wetland that may be partially the result of nitrification of the very small amount of ammonia that enters. However, individual profiles vary greatly, and it would be misleading to attempt model calibration from any one profile.

Longitudinal profiles may be used to test the validity of alternative modeling assumptions. For instance, the PFD model forecasts the concentration profile through the wetland to be given by

$$\frac{C(x)}{C_{\mathrm{i}}}=\frac{2\exp\left(\frac{xPe}{2}\right)\left[(1+b)\exp\left(\frac{+bPe(1-x)}{2}\right)-(1-b)\exp\left(\frac{-bPe(1-x)}{2}\right)\right]}{(1+b)^2\exp\left(\frac{bPe}{2}\right)-(1-b)^2\exp\left(\frac{-bPe}{2}\right)} \tag{6.54}$$

where

x = distance in flow direction, wetland, m
$C(x)$ = concentration at length x, g/m^3

Interestingly, this predicted profile has not been examined in any of the literature pertaining to applications to ponds or wetlands. However, in a treatment wetland, this model conceives of swirls, which cannot move back into the inlet distribution works, nor move forward into the outlet collection works. Consequently, the longitudinal concentration profile is predicted to display an instantaneous drop at the wetland inlet. For D/uL = 0.2 and $k\tau/h$ = 3, the decrease at the inlet is 30%. This unrealistically large concentration drop has not been observed in practice, and hence the PFD model is not an acceptable alternative.

Mixtures, Weathering, and the *P-k-C** Model

Equation 6.50 represents the reduction of a single compound on transit through a treatment wetland. However, many contaminants are, in fact, mixtures. In almost all instances, water quality parameters are measured by procedures that lump individual chemical compounds into an overall or total concentration for that class of materials. BOD and TSS are examples of such lumping. It is clear that the individual components of such mixtures may be degraded or removed at different rates, and that there is a corresponding difference in removal rate constants (Crites and Tchobanoglous, 1998; Tchobanoglous *et al.*, 2000; Shepherd *et al.*, 2001; Kadlec, 2003a). There is, therefore, a distribution of rate constants across the various mass fractions of the mixture. Such a distribution may be discrete, in the case of a countable and very small number of individual compounds, or it may be continuous in the case of a very large and possibly uncountable number of constituents. Combinations of both types of distribution are also common, such as for total nitrogen (TN). TN consists of a few separately identifiable compounds (nitrate, ammonia) and lumped classes of compounds (particulate N, organic N). Total phosphorus (TP) comprises particulate (PP), dissolved organic (DOP), and soluble reactive (SRP) forms. As water containing such a mixture passes through the wetland, its composition changes because different fractions of the mixture are reduced at different rates. The mixture becomes *weathered*, a term coined to describe the selective stripping of light volatile materials upon exposure to outdoor environments.

Each fraction of the lumped material will, in general, possess its own k-value. Therefore, there is a distribution of k-values, designated by $f(k)$:

$$f(k)dk = \text{mass fraction of material with rate constant in the range } k \text{ to } k + dk \tag{6.55}$$

This k-value frequency distribution across the mass fractions of the lumped material is termed the kVD. It may also be shown for gamma distributions of k-values (Kadlec, 2003a) that the average k-value at any time during the reduction process is

$$k = \frac{k_i}{(1+\beta t)^n} \tag{6.56}$$

where

k = rate constant during the weathering process, m/yr
k_i = inlet rate constant, m/yr
n = mixture k-value distribution breadth parameter, dimensionless
t = length of time the mixture has weathered in the wetland, d
β = mixture k-value distribution weathering parameter, d^{-1}

In the wetland environment, the DVD and the kVD interact to produce the overall observed reduction in a lumped category of pollutants. However, batch testing eliminates the DVD effect as there is no distribution function for batch time. The DVD effect is also removed in theory for the (unachievable) ideal of true plug flow.

It has been noted that observed weathering behavior in real wetland situations may be represented by the TIS model Equation 6.50, wherein the parameter values are relaxed to become fitting parameters (Kadlec, 2003a). The *relaxed TIS concentration model* is, therefore, defined to be

$$\left(\frac{C-C^*}{C_i-C^*}\right) = \frac{1}{(1+k/Pq)^P} = \frac{1}{(1+k_V\tau/P)^P} \tag{6.57}$$

TABLE 6.3
Apparent TIS Numbers (*P*-Value) for Hypothetical Double Continuous Distributions

	$n = 1$	$n = 4$	$n = 8$	$n = \infty$
$N = 1.5$	0.54	0.99	1.13	1.50
$N = 2$	0.68	1.37	1.62	2.00
$N = 4$	0.86	2.19	2.86	4.00
$N = 8$	0.95	2.90	4.29	8.00
$N = \infty$	1.00	4.00	8.00	—

Note: The relaxed TIS model has been fit to doubly distributed behavior, with N stirred tanks and a kVD parameter of n.

where

k = modified first-order areal rate constant, m/d
k_V = modified first-order volumetric rate constant, d^{-1}
P = apparent number of TIS

A gamma DTD/kVD model was used to generate concentration profiles for an incoming $k_o = 0.2$ d^{-1}, and ranges $1 \le n \le \infty$ and $1.5 \le N \le \infty$ (Table 6.3). That value of k_o is appropriate for nutrients, such as TP or TN. That computer data was then fit with Equation 6.57, generating k- and P-values for each profile. The value of the rate constant was constant across all distributions, $k = 0.198 \pm 0.006$ (mean ± SD). The apparent number of TIS varied systematically, but always $P \le N$ and $P \le n$ (Table 6.3). In other words, the presence of a weathering mixture will cause a reduction in the N-value determined from an inert tracer experiment.

To illustrate the effect of model choice, the progression of concentration reduction along a flow direction is compared for plug flow, NTIS, and PTIS models (Figure 6.20). Plug flow produces a linear decline on the semi-logarithmic plot for first-order disappearance kinetics. If the hydraulics are represented by four TIS, in a tracer test, and the contaminant experiences first-order decay, the decline is no longer linear, because rates slow at longer detention time because

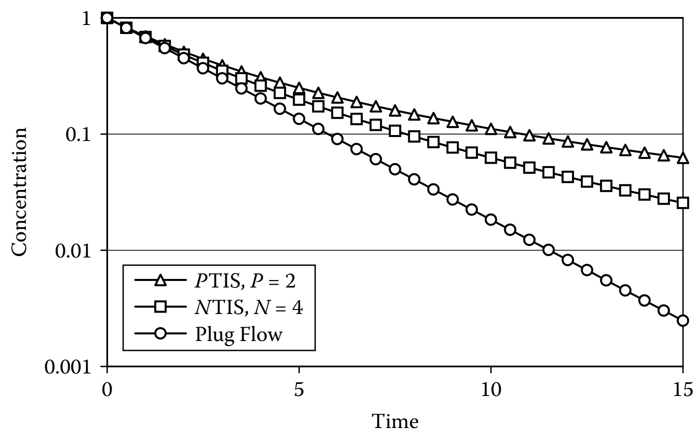


FIGURE 6.20 Comparison of time progressions along a flow path for three models: plug flow, NTIS, and PTIS.

of the early exit of some fraction of the material. If, in addition, the material is a mixture that undergoes weathering, the apparent number of tanks is reduced, for instance, to PTIS = 2. There is an even greater slowing of the decline because the rate constant decreases as travel time progresses.

As seen in Figure 6.20, there is not much difference in concentration reduction among the various models for low removals, e.g., when removal is less than 50%. However, there is a very large difference when removals are in the high range, and there can be as high as a factor of 10 in concentrations when reductions are above 99%. The specific illustration of Figure 6.20 can be expanded to include a variety of NTIS or PTIS (see Figure 6.17). It is seen that the achievement of large removals, e.g., 99%, requires very large areas if the parameter P is small, e.g., $P = 1$ or 2. This large sensitivity to the P-value implies that high pollutant reductions cannot be achieved unless the wetland is hydraulically very efficient.

The first "half" of a wetland performance model is deterministic, and is represented by Equation 6.57. This representation of the P-k-C^* model would be used to describe the central tendency (average performance) of the wetland reactor. Probability distributions represent the second "half" of the mathematical model, which in total is written as

$$C = C^* + \frac{(C_i - C^*)}{(1 + k/Pq)^P} + E \qquad (6.58)$$

The random part of the outlet time series is typically given by the intrasystem probability distribution graph or table.

Variability information may also be configured as a multiplier on the deterministic portion of the prediction. Kadlec and Knight (1996) provide such monthly multipliers for the 100% probability for undetrended data for the limited data then available, mostly for lightly loaded wetlands:

$$C(P_\omega = 100\%) = C_{\text{model}} \times F(P_\omega = 100\%) \qquad (6.59)$$

where
C_{model} = deterministic model concentration, mg/L
F = model multiplier to meet probability criterion, unitless
P_ω = probability that exceedances of frequency ω will not occur
ω = frequency of data averaging (e.g., weekly, monthly, etc.)

It is important to note that the probabilistic portion of the wetland performance model is not a "safety factor," as utilized in some wetland design procedures (Water Environment Federation, 2001). In the usual sense of the term, a safety factor provides extra capacity in design to accommodate unforeseen events and phenomena. The stochastic variability that exists in all treatment wetland outlet concentration data is not unforeseeable. In fact, these probabilistic variations are just as quantifiable as the deterministic variations caused by changes in detention time, and must be accounted in design apart from any considerations of safety factors. It is simply unacceptable to ignore this half of wetland behavior in design, as is the case in U.S. EPA (2000a) and other sources. It is possible to account for probabilistic variations in wetland performance even when using simple loading chart relationships, and this has been done in the wetland literature (Wallace and Knight, 2006; WERF database, 2006).

In subsequent chapters, the multipliers on the seasonal trend are presented for each of the common contaminants and the various wetland types. The random variation, E, can be captured through the use of a multiplier on the trend value, C_{trend}:

$$E = C_{\text{trend}} \Psi \qquad (6.60)$$

$$C = C_{\text{trend}}(1 + \Psi) \qquad (6.61)$$

where
Ψ = stochastic portion of C/C_{trend}, dimensionless

Methods for incorporating stochastic modeling in design are discussed in Part II of this book.

Rate Constant Distributions

The data from any one wetland may be used to calibrate the PTIS model. Then, the calibrations from a number of such wetlands may be used to determine the frequency distribution of those k-values. Here, the reduction of ammonia in FWS wetlands is used as an example. In Chapter 9, this aspect of nitrogen removal is dealt with in much more detail. Here, the apparent removal of ammonia is used as an illustration of the quantification of frequency distributions of parameter values for wetland models. For purposes of illustration, systems that produce ammonia from mineralization of organic nitrogen are excluded. Often, the N-value is not known, because a tracer test has not been run. So, for illustration purposes, it will be assumed that $P = N = 4$, which is a mean N-value for FWS wetlands (see Table 6.1). The value of C^* is presumptively taken to be zero in this analysis. Equation 6.57 may be used to calculate a long-term average k-value from the hydraulic loading as well as the inlet and outlet concentrations averaged over the period of record. By this procedure, temperature effects are lumped into the variability, although it is known that water temperature will be a significant component of the set of conditions which lead to the variability. Each wetland is accorded one value for its entire period of record, thus averaging over a number of annual periods that differs for each wetland.

The probability distribution of these k-values for a set of 131 FWS wetlands is quite broad (Figure 6.21). The mean is $k = 18$ m/yr, but the distribution contains some very high rate constants. Accordingly, the median $k = 11.5$ m/yr. However, the range is 0.8–308 m/yr, and the SE of the mean is 3 m/yr (SD = 30 m/yr). Clearly, this distribution is too broad to give much confidence in design for a mean k-value. It is obviously necessary to understand the components of the wetland environments and layouts that contribute to either

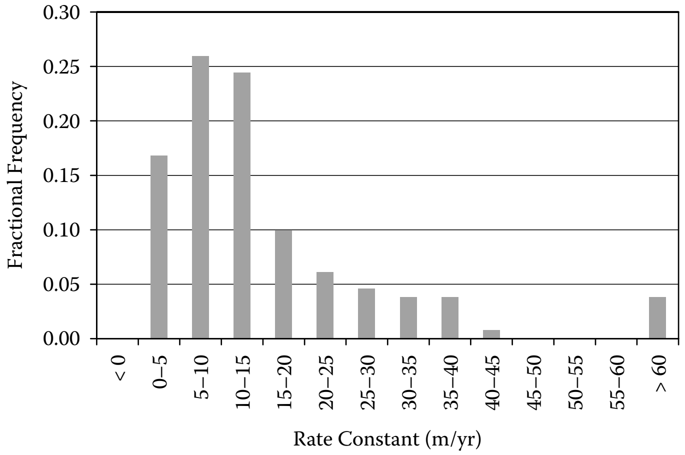


FIGURE 6.21 Rate constants for ammonia removal in FWS wetlands. The presumption is for 4 TIS. Ammonia-producing systems are excluded. Each of the 131 wetlands is represented by one point, representing its entire period of record. The data were not temperature-corrected.

high or low *k*-values. Very importantly, it would be exceedingly dangerous to place great trust in any one wetland as a prototype for all others. In addition, it would be pure coincidence if any new wetland were to behave as the mean of the distribution.

As a result of these considerations, recommendations for design must go beyond the concept of a universal, or average, *k*-value that may be used for any wetland.

Data Fitting

Equation 6.57 represents an alternative for quantifying wetland performance data. It has three potentially adjustable parameters, C^*, k, and P. Tracer information gives an upper bound for P. The value of C^* does not have to accommodate both hydraulic and biogeochemical plateau effects for long detention times. The value of P is a free-fitting parameter, subject to the constraint of $P < N$, where N is the tracer TIS number. The value of C^* represents the only biogeochemical background, because speciation effects have been removed to the parameter P. C^* may be selected in one of two ways. It may be considered a free parameter, constrained by $C^* > 0$, or it may be selected to be the lowest concentration ever measured in a comparable situation, such as at far down-gradient locations in impacted pristine systems.

The best procedure would be to fit a three-parameter model to the data, adjusting k, C^*, and P. However, if the available data cover only a small reduction in the inlet concentration from values well above C^*, there is not sufficient information to gain a good estimate of C^*. Conversely, if most of the data are in the region near C^*, a good estimate of the k-value is not possible. These results suggest:

1. For high inlet concentrations ($C_i >> C^*$), it is better to guess C^* and gain good estimates of k and P.
2. For low inlet concentrations ($C_i < 3C^*$), it is better to guess $P < N$, and gain good estimates of k and C^*.

It is not as onerous as it may seem to independently estimate C^*. Data from a wide range of treatment wetlands suggest that virtually all individual chemicals have zero wetland background levels. Exceptions include BOD, COD, organic nitrogen, and pathogens, as will be discussed in subsequent chapters.

Often, the worst fitting procedure is to choose $C^* = 0$, and select $N = \infty$, which is the plug flow model.

Synoptic Error

There will typically be a set of contemporaneous values of wetland inlet and outlet concentrations, together with accompanying flows and other information. Calibration consists of selection of rate parameters that minimizes the error between those field observations and the calculated model values from Equation 6.57 or an equivalent.

It is clear that paired contemporaneous measurements of input and output flows and concentrations have little chance of providing a quantitatively accurate description of removals, because of the transport delay in the wetland. Typical hydraulic detention times are of the order of several days to more than a week, which implies a significant shift in event timing if the wetland is in plug flow. However, tracer testing of hundreds of treatment wetlands has shown conclusively that no treatment wetland exhibits plug flow; rather, the detention time distribution extends to three or more nominal detention times (Kadlec, 1994a). Therefore, there are remnant effects of inlet events at the outlet after three or more nominal detention times. The only chance of avoiding a transport delay artifact (synoptic error) is to compare inlet and outlet measurements averaged over more than those three nominal detention times.

The water leaving the wetland may have entered anywhere from about a tenth to three or four times the nominal detention time earlier. If there is a time series of changing inlet concentrations or flows, instantaneous, contemporaneous inlet–outlet data should not be used to calibrate the model. Rather, time averages over at least three detention times should be used.

6.7 OTHER FACTORS AFFECTING TREATMENT PERFORMANCE

Definition of the Rate Constant

It is seen that the Damköhler number ($Da = k/q = k\tau/h$) grouping is common to many models, including the TIS and PFD models. There are differences in the interpretation of both h and τ, as well as different assumptions concerning the possible relations among k, h, q, and τ.

Which Detention Time?

As seen in Chapter 2 and the discussion of tracer testing in this chapter, there may be considerable difference between the nominal detention time and the mean tracer (actual) detention time, exemplified by the volumetric efficiency, e_V. Tables 6.1 and 6.2 show that on average, $\tau_{actual} = 0.82\ \tau_n$ for a sampling of FWS wetlands, and $\tau_{actual} = 0.91\ \tau_n$ for a sampling of SSF wetlands. During data analysis, either may be used, if a tracer test is available. The resultant calibrated values of k will differ markedly, depending upon which alternative is chosen. In this book, the nominal detention time is chosen, because tracer testing may not be available to provide the actual detention time. This implies that the volumetric efficiency is absorbed into the rate constant.

Areal versus Volumetric Rate Constants

A good portion of the treatment wetland literature (Crites and Tchobanoglous, 1998; U.S. EPA, 1999; Water Environment Federation, 2001) utilizes a volumetric rate constant, defined as:

$$k_V = \frac{k}{e_V h} \tag{6.62}$$

Those sources also estimate values of $0.65 < e_V < 0.75$, which are referred to as "porosity" (see Chapter 2). Consequently, k_V-values from those sources are based on such e_V-values.

Another fundamental difference between the use of k and k_V arises from the assumption of constancy of k-values or k_V-values, as follows:

- If k_V is assumed constant, deeper water (greater h) provides more detention time, without any penalty of reduction of the value of Da for the system.
- If k is assumed constant, deeper water (greater h) provides more detention time, but with a penalty of reduction of the value of Da for the system which compensates for, and removes the advantage of, the greater deep water detention.

Side-by-side studies have been conducted for both FWS and HSSF wetlands, in an effort to investigate the effect of depth increases at constant flow. Analyses of data from FWS studies at Arcata, California, show that k was nearly constant, and therefore k_V decreased with increasing depth (data in Gearheart *et al.*, 1983). As the depth, and hence detention, increased, the calculated k_V decreased proportionately for BOD and fecal coliforms (Table 6.4). Side-by-side studies of HSSF wetlands at Les Franqueses del Valles, near Barcelona, Spain (García, 2003; García *et al.*, 2004b), produced removals that increased with decreasing depth (Table 6.5), thereby indicating that k_V-values would also increase with decreasing depth. Indeed, the García *et al.* (2004b) data produced *areal* k-values that increased with decreasing depth, ranging 22–33 m/yr for 27 cm depth, and 4–7 m/yr for 50 cm depth. García *et al.* (2004b) attributed these effects to differences in the chemical environment created by shallower water, in particular, the occurrence of denitrification in shallow systems.

TABLE 6.4
Plug Flow Removal Rate Constants in Side-by-Side FWS Wetlands Operated at the Same Hydraulic Loadings

Flow (m^3/d)	Depth (m)	Increase in HRT (%)	BOD PF Rate Constant (d^{-1})	Decrease in BOD k_V (%)	Fecal Coliform PF Rate Constant (d^{-1})	Decrease in FC k_V (%)
93	0.40	—	0.29	—	1.33	—
94	0.55	37	0.17	42	0.62	53
86	0.36	—	0.25	—	0.83	—
83	0.61	76	0.13	49	0.51	38
45	0.30	—	0.28	—	0.83	—
49	0.49	49	0.14	48	0.41	51
29	0.33	—	0.14	—	0.36	—
29	0.53	78	0.08	40	0.32	10
23	0.35	—	0.14	—	0.53	—
24	0.50	39	0.09	36	0.38	28

Source: Data from Gearheart *et al.* (1983) *City of Arcata Marsh Pilot Project, effluent quality results—system design and management*. Final Report to the North Coast Regional Water Quality Board (Santa Rosa, California) and State Water Resources Board (Sacramento, California).

TABLE 6.5
Percent Removals in Side-by-Side SSF Wetlands Operated at the Same Hydraulic Loadings

Parameter	Unit	Deep (50 cm)	Shallow (27 cm)
COD	mg/L	63.5 ± 1.4	74.5 ± 6.4
BOD_5	mg/L	56.5 ± 2.7	77.5 ± 9.2
Ammonia	mg/L	26.5 ± 2.3	44.5 ± 9.2
Dissolved Reactive Phosphorus	mg/L	5.2 ± 3.1	16.0 ± 8.5
Relative HLR	—	1.00	1.00
Relative HRT	—	1.00	0.54

Note: The shallow systems do much better, despite the fact that they have less detention time.

Source: Data from García *et al.* (2004b) *Ecological Engineering* 23(3): 177–187.

In FWS wetlands, the biogeochemical processes, which remove and sequester pollutants, are closely associated with plants, biofilms, and sediment interfaces. In total, these represent a "biomachine" that processes contaminants and nutrients. Such action zones are typically apportioned to wetland area to a greater extent than to wetland water volume. Thus, when the wetland area is doubled at constant depth, the detention time is doubled, and the biomachine is doubled. But, when the water depth is doubled at constant area, there are not more plants, and biofilms and interfaces do not necessarily increase in proportion to water depth (Figure 6.22). In fact, it is often not possible to deepen the water and retain the same ecology, because of the hydropattern requirements of vegetation (see Chapter 3).

In contrast, properly designed SSF wetlands can support vegetation at virtually any water depth, because the plants are situated on top of the media and root into the water just below the surface. The media also provide the majority of the interfacial and biofilm area, which is in proportion to water depth. However, plant roots typically do not penetrate more than 20–30 cm, as discussed earlier in this chapter. There is debate over how much effect plants and their roots have on treatment potential, ranging from the suggestion that they completely control the degree of treatment (Reed *et al.*, 1995) to the idea that they have no effect at all (Langergraber, 2001).

Either k or k_V can be used to represent a data set or be used in design. However, the use of k_V for FWS or HSSF wetlands requires the accompanying information on water depth (h) in the sizing equations because of the high probability of important depth dependence. This depth dependence usually means that more detention time created by deeper water is counteracted by a decrease in the volumetric rate constant. Conversely, data indicate that (the areal) k is nearly independent of depth. Nonetheless, data analysis and design require knowledge of the water depth, because of the implied changes in the type of vegetation that may be fostered and maintained.

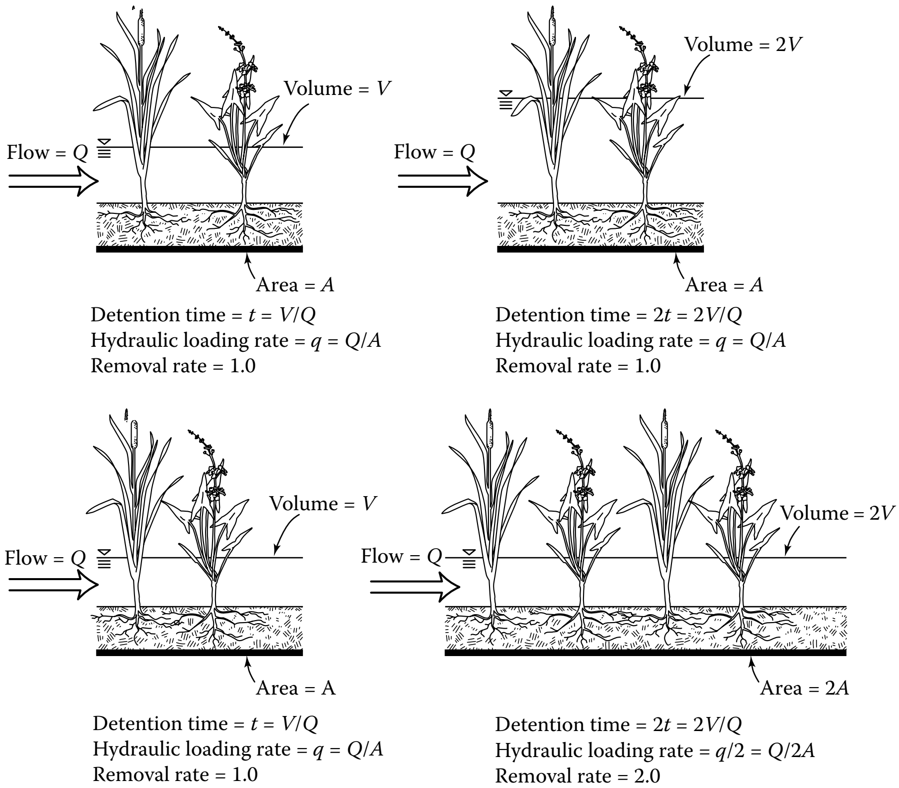


FIGURE 6.22 Conceptual distinction between increasing detention time with deeper water and with more area.

Temperature and Season

Temperature effects on k or k_V have often been summarized by use of the modified Arrhenius equation:

$$k_T = k_{20}\,\theta^{(T-20)} \tag{6.63}$$

where k_T is the rate constant at temperature $T = T$°C, and k_{20} is the rate constant at 20°C. Values of the temperature correction factor (θ) may be estimated for data sets with adequate operational temperature data. However, it should be noted that incorrect historical presumptions still pervade the treatment wetland literature.

Microbial processes have been studied in several types of "conventional" wastewater treatment devices, including activated sludge plants and trickling filters. The Arrhenius factors for some contaminant reductions have been documented for those situations (see, e.g., Metcalf and Eddy, Inc., 1991). This prior information for other technologies lead early wetland workers to assume that the temperature coefficients would be the same for wetlands, which they viewed as microbial reactors. For instance, a BOD temperature coefficient of θ = 1.06 was *assumed* by Reed *et al.* (1995), and has been propagated in other subsequent publications (Crites and Tchobanoglous, 1998; Water Environment Federation, 2001; Crites *et al.*, 2006). However, examination of treatment wetland data (and other natural system data) does not yield such a value for BOD (Kadlec and Reddy, 2001). For 23 FWS wetlands, they found an average θ = 0.983; for three ponds, θ = 1.005; and for two overland flow systems, θ = 1.012. The situation in wetlands is apparently more complicated than just microbial processing.

Many of the variables that go into a mechanistic BOD model are temperature-dependent, such as diffusion coefficients and the biofilm rate constant (k_b). However, the apparent rate constant (k) is a combination of those parent variables, and therefore exhibits a different temperature dependence. The theta model is used here to explore the consequences. For illustration, assume:

1. The rate constant $k_b = k_{b20}\,\theta^{(T-20)}$. The value of θ = 1.05, after Polprasert and Agarwalla (1994).
2. The diffusion coefficients have $\theta_w = 1.025$ and $\theta_b = 1.00$, after Polprasert and Agarwalla (1994).
3. The relative amounts of biofilm surfaces are greater in winter, after litterfall has occurred. Assume there is 25% more litter during winter than spring, and 25% less litter during summer.
4. Climatic conditions give a winter temperature of 5°C and a summer temperature of 25°C.
5. The values of the 20°C parameters are those determined by Polprasert and Agarwalla (1994).

The resulting θ-value for the overall first-order areal uptake coefficient (k) is then 0.999. The temperature dependencies cancel each other.

The situation for nutrients is even more complex because light nutrient loadings are strongly influenced by plant and algal uptake on a seasonal basis. Microbial activity follows an annual pattern with a peak in midsummer, but uptake peaks earlier, in spring (Figure 6.23). To the extent that growth has first claim on nutrients, removal will be out of phase with the annual temperature cycle. The loading region sensitive to growth uptake has been postulated to be about 120 g/m²·yr for nitrogen (Kadlec, 2005d). Phosphorus removal would be expected to be in phase with growth, since microbial conversions do not remove total phosphorus. When growth effects are dominant, the k-values change seasonally:

$$k = k_j \quad j = 1, 2, 3, \ldots 12 \tag{6.64}$$

where j is the month number. Although higher frequency might be used for the seasonal change pattern, monthly values will be sufficient in cases where detention time lags are on the order of a week.

Variability and Data Folding

Many factors contribute to random variability in the outlet concentrations from treatment wetlands. This variability is

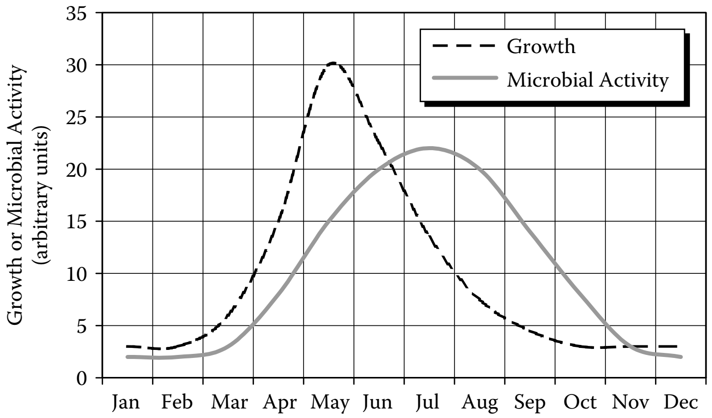


FIGURE 6.23 Annual patterns of growth and microbial activity in a temperate climate. Growth peaks in spring, when temperatures and microbial activity are at moderate levels.

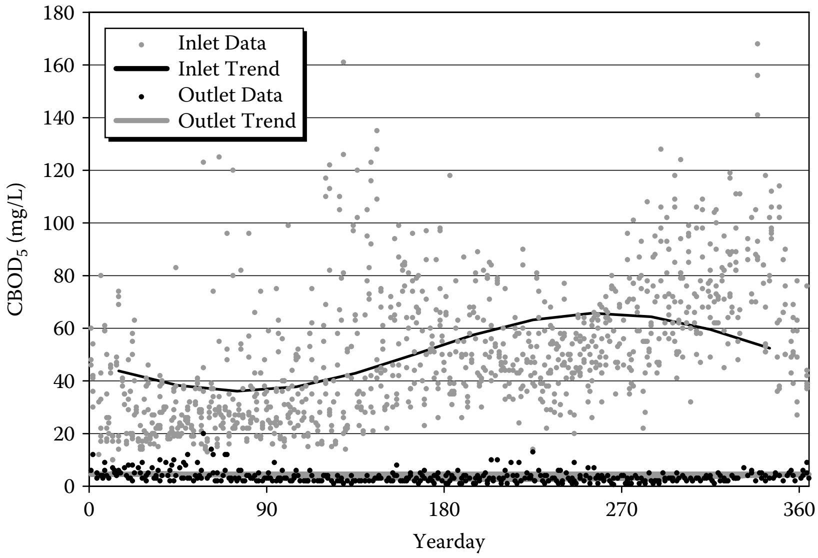


FIGURE 6.24 Annualized progression of $CBOD_5$ in the Arcata, California, wetland system. The inlet is to the treatment wetlands, and the outlet is from the enhancement wetlands. The period of record is 1992–1999. (Data from TWDB database (2000) *Treatment Wetland Database (TWDB).* Website developed for U.S. EPA. http://firehole.humboldt.edu/wetland/twdb.html. Last updated November 2000. Compiled by B. Finney. U.S. EPA: Washington, D.C.)

typically not small, with coefficients of variation of 20%–60% being common. The deterministic models presented above reproduce the central tendency of performance, but not the random variability. Whether there is microbial or vegetative control, seasonal patterns of wetland variables are the rule, accompanied by a random variable term (Kadlec, 1999a).

$$C = C_{avg}\left[1 + A \cdot \cos\left(\omega(t - t_{max})\right)\right] + E \qquad (6.65)$$

where

- A = fractional amplitude of the seasonal cycle, dimensionless
- C = instantaneous outlet concentration, mg/L
- C_{avg} = average (trend) outlet concentration, mg/L
- E = random portion of the outlet concentration, mg/L
- t = time of the year, Julian day
- t_{max} = time of the year for the maximum outlet concentration, Julian day

The deterministic portion of this representation may also be modeled by the *k*-rate technique with appropriate rate constants and background concentrations, both of which may respond to temperature and season. The stochastic portion (E) will have a probability distribution, which will be different depending upon sampling frequency and sample averaging period.

The existence of the error term (E) means that sampling must either be at high frequency, or cover many annual cycles before meaningful trend averages can be determined. Data from several years may be “folded” to create an annualized grouping, distributed across the year according to Julian day. This use of many annual cycles has the advantage of including year-to-year variations in climate, flow, and ecosystem condition.

As an example, consider the $CBOD_5$ behavior of the Arcata, California, treatment and enhancement marshes (Figures 6.24–6.26). Weekly sampling produces a cloud of data which spans a wide range of values at any particular time of the year (Figure 6.24). Although compressed by the vertical scale required to show inlet concentrations, the variability in the outlet $CBOD_5$ is also quite high. Because the mean outlet concentration over the 1992–1999 period of record was quite low, just under 4 mg/L, there occurs a skewed distribution of departures from the trend behavior (Figure 6.25). This leads to a nearly log-normal distribution of fractional errors, which is to be expected whenever concentrations are limited on the low end by analytical accuracy near detection limits. Because of the luxury of eight years’ worth of data, it is possible to determine a *k*-value, which describes the relation between the average inlet and average outlet concentrations. The *P*-*k*-*C** model is well calibrated by $P = 8$, $C^* = 3$ mg/L, and $k = 57$ m/yr (Figure 6.26). The high value of P is not unexpected, because there are five wetlands in series in the Arcata system. Thus each wetland unit behaves like less than two tanks in series.

Water Losses and Gains

In general, literature values of rate constants have not been corrected for water losses and gains. In some instances, water budget information was not collected; in other cases atmospheric losses and gains were not significant. Therefore, water mass balance effects are the cause of some fraction of the variability in rate constant data.

It is readily possible to adjust the *P*TIS model to account for excessive rain or evapotranspiration. The wetland is broken into P segments, corresponding to the model (see Figure 6.18). First, the water mass balance for the first

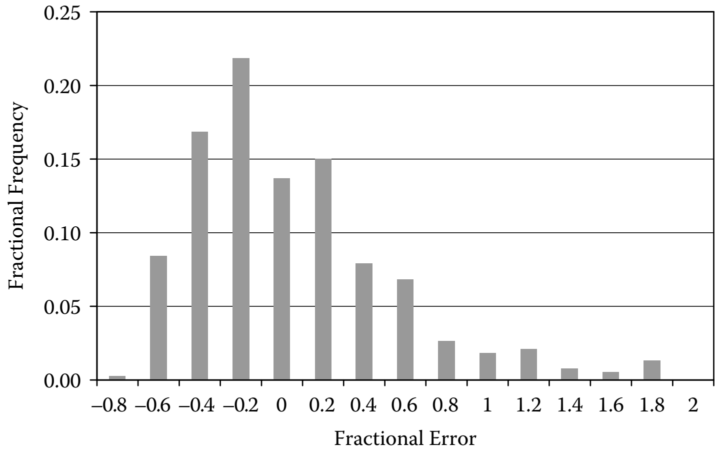


FIGURE 6.25 Distribution of departures from the annual trend for $CBOD_5$ at the Arcata, California, treatment wetland. The period of record is 1992–1999. (Data from TWDB database (2000) *Treatment Wetland Database (TWDB)*. Website developed for U.S. EPA. http://firehole.humboldt.edu/wetland/twdb.html. Last updated November 2000. Compiled by B. Finney. U.S. EPA: Washington, D.C.)

of the wetland segments, designated by the subscript "1," for steady-state, nonuniform flow is

$$Q_1 = Q_i + A_1 \cdot (P - ET - I) \tag{6.66}$$

where

A_1 = area of the first segment (tank), m^2
ET = evapotranspiration, m/d
I = infiltration, m/d
P = precipitation, m/d
Q_i = inlet flow rate, m^3/d
Q_1 = outlet flow rate from segment #1, m^3/d

The pollutant mass balance for the same first segment, for steady-state, nonuniform flow is:

$$Q_1C_1 = Q_iC_i - (I \cdot A_1C_1) - (k \cdot A_1 \cdot (C_1 - C^*)) \tag{6.67}$$

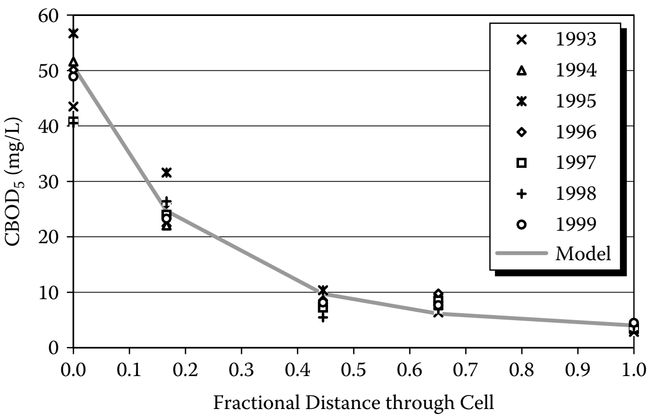


FIGURE 6.26 Progression of $CBOD_5$ through the Arcata, California, treatment wetland complex, including both treatment and enhancement wetland systems. The period of record is 1992–1999. (Data from TWDB database (2000) *Treatment Wetland Database (TWDB)*. Website developed for U.S. EPA. http://firehole.humboldt.edu/wetland/twdb.html. Last updated November 2000. Compiled by B. Finney. U.S. EPA: Washington, D.C.)

where

C_1 = concentration out of and in the first segment (tank), mg/L
C_i = concentration into the first segment, mg/L
C^* = background concentration, mg/L

Here, rainfall has been assumed to have a zero pollutant concentration, and infiltration is assumed to occur at the outlet concentration. Combining Equation 6.66 with Equation 6.67 gives the concentration exiting hypothetical segment number one:

$$C_1 = \frac{Q_iC_i + (k \cdot A_1 \cdot C^*)}{Q_i + (A_1 \cdot (P - ET)) + (k \cdot A_1)} \tag{6.68}$$

This computation is then repeated for the remaining segments, in each case using the outlet concentrations and flows from the preceding unit. The wetland outlet concentration is that exiting from the final hypothetical segment.

Interactions with Solids

The water detention time does not appropriately reflect transport delays due to storage in or on stationary compartments of the wetland: the substrate, sediments, and biota. The contaminant residence time will vary markedly depending on how many times it has been "parked" and recycled in various active or passive storage compartments during its passage through the wetland.

As an example, consider the hypothetical compartmentalization and storages in a wetland for nitrogen (Figure 6.27). The overall nitrogen content of the various solid storages totaled over 200 gN/m^2. Nitrogen removal rates were of the order of 2 $gN/m^2 \cdot d$, corresponding to hydraulic detention times of about five days. Therefore, the nominal displacement time for solid phase nitrogen is about 100 days. However, first-order turnover is more likely than displacement, and not all the solid phase nitrogen is available for turnover. Reported turnover times are rapid, of the order of one day, for sorbed ammonium (Sikora *et al.*, 1995;

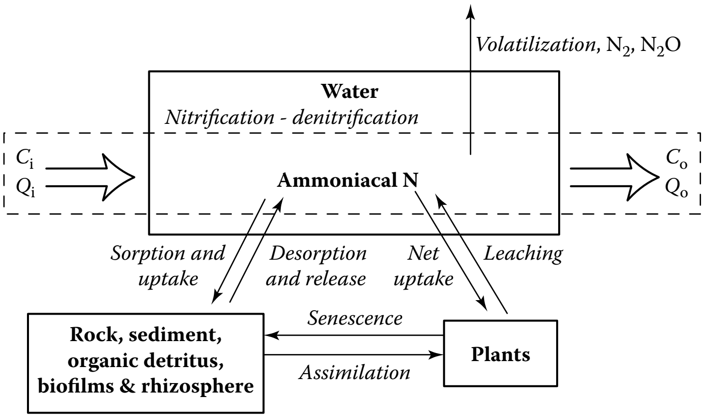


FIGURE 6.27 Simplified nitrogen processing schematic for high ammonium loading and low removal. (From Kadlec *et al.* (2005) *Ecological Engineering* 25: 365–381. Reprinted with permission.)

Tanner *et al.*, 1999). However, plant detritus turns over much more slowly, with half-lives of up to a year or more (Hietz, 1992; Tanner, 2001a). These approximations for nitrogen storage and fluxes indicate that the nitrogen detention time in the constructed wetland is far greater than the water detention time.

The importance of the large pools of solid phase nitrogen lies in the fact that small percentage changes in those storages can have large consequences for the water concentrations. For instance, the seasonal senescence of biomass is capable of adding back a portion of the nitrogen stored during the growing season (Sartoris *et al.*, 2000; Tanner, 2001a), thus adding an element of seasonality to performance.

A simplified model of nitrogen tracer (^{15}N) movements in the wetland environment is possible when some of the more minor intercompartmental transfers are ignored, and the solids compartments are lumped (Kadlec *et al.*, 2005). This model allows for unidirectional plant uptake, reversible exchange with wetland solids, and microbial conversion of ammonium to nitrogen gases. It has been calibrated to treatment of dairy wastewater with high ammonium in gravel bed mesocosms with good success ($R^2 = 0.97$–0.99) (Tanner *et al.*, 1999). However, that calibration was for batch operation, whereas here we are interested in the dynamics of tracer movement through a spatially variable, longitudinal SSF wetland. The plug flow spatial and dynamic equations are:

$$\varepsilon \frac{\partial C}{\partial t} + u \frac{\partial C}{\partial y} = -k_{\mathrm{Vu}} C + k_{\mathrm{Vr}} X - k_{\mathrm{Vg}} C \qquad (6.69)$$

$$(1-\varepsilon) \frac{\partial X}{\partial t} = k_{\mathrm{Vu}} C - k_{\mathrm{Vr}} X \qquad (6.70)$$

where

C = water phase concentration, g/m^3
k_{Vg} = growth and conversion rate constant, d^{-1}
k_{Vr} = release rate constant, d^{-1}
k_{Vu} = uptake rate constant, d^{-1}
t = time, d
u = water velocity, m/d
X = solid phase concentration, g/m^3
y = distance, m
ε = porosity

It should be noted that a volumetric solid phase concentration (X) has been used. It is related to the commonly used mass concentration by

$$X = \rho \cdot C_{\mathrm{S}} \qquad (6.71)$$

where

C_{S} = solid phase concentration, g/kg
ρ = solid density, kg/m^3

The model may be implemented in a sequence of segments, thus converting it to a PTIS model. Figure 6.28 presents rough calibrations to the field scale wetlands for $P = 5$. About 85% of the incoming tagged ammonia was not converted, and ultimately exited the system. Only a tiny portion of the tagged ammonium nitrogen goes directly to the wetland exit (<1%). The ^{15}N detention time for the field scale wetlands is about 50 days, or 12 times the water detention time.

The correct concept for the longitudinal transport of interactive substances, such as nitrogen and phosphorus, includes major interactions with wetland solid compartments. Transport with water is rapid, but the exiting compounds originated from storage, and not from the water that entered only one or a few detention times earlier. Pulses of incoming nutrients must work their way through solid storages. Consequently, there is a second type of synoptic error that must be accounted, which results from the wetland storage delay.

System Start-Up

The models discussed in the previous text pertain to fully developed treatment wetlands. Plants are ordinarily introduced into constructed treatment wetlands as seeds or propagules at biomass densities far less than those that will ultimately develop. This process of grow-in typically takes a period of one to three years, depending upon climate. The amounts of nutrients taken up during the initial grow-in are substantial in many cases, and represent a removal process that is not sustainable in the long term. Development of a litter layer takes even longer. Because submerged plant material creates attachment substrate for microbes, and a carbon source for metabolism for some of them, the wetland may not

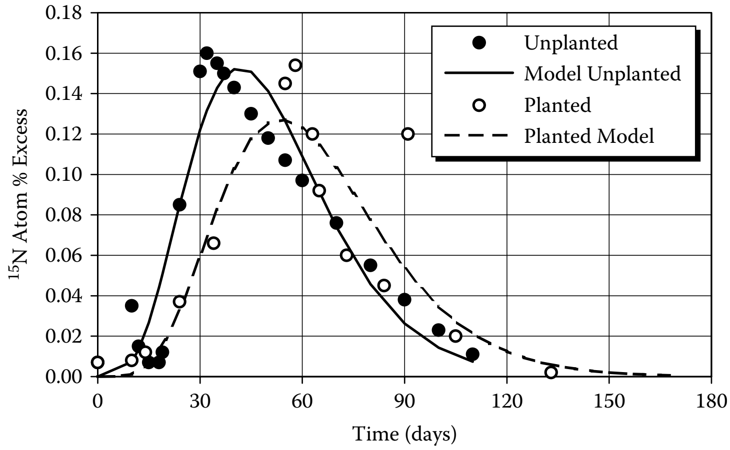


FIGURE 6.28 Response of planted and unplanted SSF wetlands to an impulse of $^{15}NH_4$. The water detention time in these systems was about five days. (From Kadlec *et al.* (2005) *Ecological Engineering* 25: 365–381. Reprinted with permission.)

be fully functional during start-up. Nonnutrient chemicals are likely to be less affected, but processes such as sorption still require the full ecosystem to reach their potential.

These large ecosystem start-up effects can dominate nutrient removal for many months (Busnardo *et al.*, 1992; Tanner *et al.*, 1998; Tanner, 2001a). Therefore, calibration of steady-state models during such a period will not produce rate constants that are representative of long-term sustainable operation.

Figure 6.29 shows an illustration of start-up effects for an aerated HSSF designed to remove ammonia from landfill leachate (Nivala, 2005). Upon implementation of wetland aeration, it took the microbial community approximately six months (under winter operating conditions) to adapt to the ammonia loading.

6.8 DANGERS OF EXTRAPOLATING WETLAND PERFORMANCE DATA

The models discussed in the previous text are all quite simplistic, in that they purport to describe general features of complicated processes. Many ecosystem compartments are lumped together, and the models are therefore termed "highly aggregated." The decision to use highly aggregated design models (all those under consideration fit that description) carries the implied penalty of great risk in extrapolation beyond the calibration conditions. The safest criterion for extrapolation is to not do it; only interpolation on the calibration sets should be allowed. A less conservative approach would be to avoid extrapolation to conditions that are known to threaten the integrity of the vegetative community that typified the calibration. In any case,

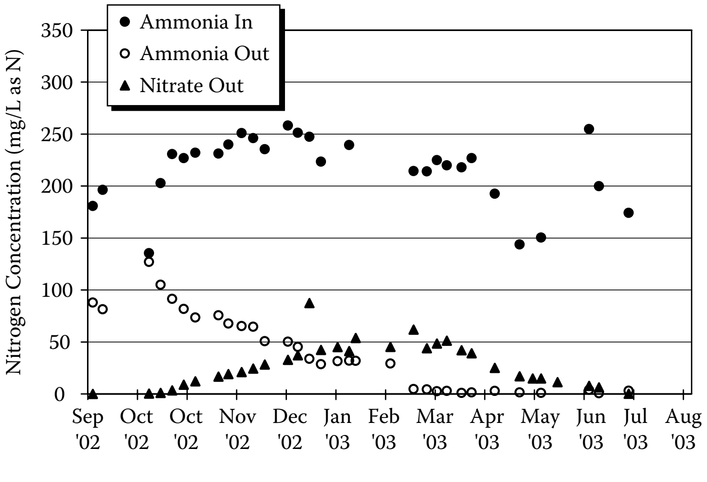


FIGURE 6.29 Example of start-up effects for ammonia removal in a HSSF wetland. This wetland is mechanically aerated to remove oxygen transfer limitations. Note that effluent ammonia concentrations are declining despite lower water temperatures over the winter months. The hydraulic loading rate was relatively constant over the time period reported. (Data from Nivala (2005) Treatment of landfill leachate using an enhanced subsurface-flow constructed wetland. M.S. Thesis, Department of Civil and Environmental Engineering, University of Iowa, Iowa City.)

extrapolation should not be to water chemistries known to be outside the range of the calibration.

Hydraulic variables present difficulties in two ways: they may affect the performance of a specific vegetative community, and they may affect that community itself. For instance, removal performance may not differ too much from steady to pulsed operation. As long as the vegetation is not altered, we are faced only with the issue of how that ecosystem averages the event-driven environment. Depth variation may not be a large factor within certain ranges if the variations are not of long duration, but average depth is likely to be an important factor. The larger risk is that the ecological communities become affected, and change to communities that do not match the calibration. For example, cattails find it difficult to survive at long hydroperiods in deep water. SAV may suffer if hydroperiods are too short. Prolonged dry-out may pose problems under some design scenarios, or at least affect the subsequent treatment performance.

Internal flow patterns are controlled by topography, depth, and compartmentalization. Pollutant removal is known to be sensitive to those flow patterns. Therefore, the applicability of a specific TIS model is conditioned on the anticipated TIS for the extrapolation. There are very large implications, especially for high removals, as was shown in Figures 6.19 and 6.22.

Extrapolation from a wetland of one type to another is clearly not a reasonable step. The microbial communities, as well as the character and magnitude of the biogeochemical cycles, may differ markedly. Forested ecosystems will not necessarily perform with the same rate constants as emergent marshes, even if the hydrologies are comparable.

As a consequence of these considerations, it is not a good practice to use model parameters (P, k, C^*, θ) in situations outside the ranges of operating values from which they were derived. There are not "universal" values of rate constants, as offered in many literature sources (e.g., Reed *et al.*, 1995; Water Environment Federation, 2001; Crites *et al.*, 2006). Nor should the reported central tendencies of parameters, such as rate constants (Kadlec and Knight, 1996), be interpreted as "universal" values. The parameter values obtained from various operating wetland systems vary widely, depending upon factors described in preceding sections. It is therefore prudent to examine the origin of a particular calibration set (P, k, C^*, θ) before using it in design calculations. The questions to be addressed include:

- Do the inlet and outlet concentration ranges of the calibration set include the ranges to be considered in design?
- Do the ranges of detention times and hydraulic loadings of the calibration set include the ranges to be considered in design? Are the intended water depths in the calibration range?
- Is the configuration of the system under design comparable to that of the calibration wetland(s)? This includes aspect ratio, number of physical compartments, and size. Consideration should be given to the N-values (or P-values) of calibration and design systems. Further, small mesocosms may not be representative of a large system.
- Are the climatic conditions for the calibration set similar to those of the intended design? Extremes of rainfall, evapotranspiration, and temperature may affect the calibration values. For instance, the seasonal freeze-up of temperate wetlands separates them from subtropical systems.
- Is the ecology of the calibration wetland(s) comparable to that of the intended wetland under design? At a minimum, wetland types must correspond across the variants of emergent marshes, floating plants, submerged aquatic vegetation, and possibly open water fraction.

Insofar as the intended design departs from calibration conditions and information, an increasing degree of risk is engendered. Intersystem data is a valuable aid to quantifying that risk. However, many systems do not possess enough information to gain firm estimates of rate model parameters. Therefore, the risk assessment associated with transferability of available data is better accomplished via other methods of data representation, as will be described in subsequent chapters.

SUMMARY

The following circumstances often apply to treatment wetland data analysis:

1. Steady flow
2. Negligible water gains and losses
3. Concentrations averaged over several nominal detention times

The central tendencies of wetland outlet concentrations are then often well represented by the PkC^* model:

$$\frac{C - C^*}{C_i - C^*} = \frac{1}{(1 + k/Pq)^P}$$

for microbial control (high loading) $\qquad k_T = k_{20}\theta^{(T-20)}$

for vegetative control (low loading) $\qquad k = k_j \qquad j = 1, 2, 3, \ldots 12$

The k-values for a group of similar wetlands will display a frequency distribution. Choice of a high or low value from such a distribution for design purposes requires assessment of the desired degree of risk and other wetland factors.

This is the TIS model, with a modified number of tanks $P \le N$, where N = the tracer-determined number of tanks. For a single chemical compound in a mixed batch system, or for long narrow wetlands with many compartments, $P = N \rightarrow \infty$, and the exponential form may be applied.

There are quantifiable excursions around this central tendency that can be included as a stochastic addition:

$$C = C^* + \left(\frac{(C_i - C^*)}{(1 + k/Pq)^P} \right) + E$$

Variability information may also be configured as a multiplier on the deterministic portion of the prediction.

$$C = C_{trend}(1 + \Psi)$$

where $(1 + \Psi)$ is the multiplier that creates the concentration exceeded not more than a specified fraction of the time.

7 Suspended Solids

A major function performed by wetland ecosystems is the removal of suspended sediments from water moving through the wetland. These removals are the end result of a complicated set of internal processes, including the production of transportable solids by wetland biota.

Low water velocities, coupled with the presence of plant litter (in FWS wetlands) or sand/gravel media (in HSSF and VF wetlands), promote settling and interception of solid materials. This transfer of suspended solids from the water to the wetland sediment bed has important consequences for the quality of the water, as well as the properties and function of the wetland ecosystem. Many pollutants are associated with the incoming suspended matter, such as metals and organic chemicals, which partition strongly to suspended matter. In FWS wetlands used for municipal wastewater treatment, the accretion of solids contributes to a gradual increase in the bottom elevation of the wetland. However, wetlands used to treat urban or agricultural stormwater, or those exposed to periodic ancillary flooding, may have rapid accretions in the inlet zone.

In HSSF and VF wetlands, incoming suspended matter is removed primarily through the mechanisms of interception and settling. Although particle resuspension due to wind, wave, or animal activity can play an important role in the sediment cycle of FWS wetlands, these mechanisms are minimized in HSSF and VF wetland systems. As a result, particulate matter tends to accumulate in HSSF and VF wetland beds, with profound consequences on hydraulic conductivity and system performance.

It should be noted that the concept of using VF filter beds to remove incoming total suspended solids (TSS) as the initial stage of a treatment process dates back to the 1960s. This concept originated with Dr. Kathe Seidel, and came to be known as the Max Planck Institute Process (MPIP) or Krefeld Process (Seidel, 1966; Liénard *et al*., 1990; Brix, 1994d; Börner *et al*., 1998). The MPIP system consisted of batch-fed vertical flow wetland beds followed by HSSF wetland stages for further effluent polishing.

7.1 SOLIDS MEASUREMENT

TSS are measured gravimetrically after filtration and drying (Method 2540D; APHA, 1998), and reported in mg/L. The organic content is characterized as volatile suspended solids (VSS), determined from the weight loss on ignition at 550°C. The TSS method has been subjected to considerable criticism by Gray *et al*. (2000) for use on "natural" waters, and these authors recommend a suspended sediment concentration (SSC) analysis as a replacement (Method D 3977.97; ASTM, 2000). One fundamental difficulty is the representativeness of aliquots, especially if they contain sand particles. A second difficulty is the wide variability of the TSS method in low concentration ranges. Gray *et al*. (2000) quote the Standard Methods precision as a 33% coefficient of variation at 15 mg/L. TSS measurements are likely to be biased low compared to SSC measurements.

Turbidity in water is caused primarily by suspended matter, although soluble colored organic compounds can contribute. Therefore, turbidity is sometimes used as a surrogate for gravimetric measurement of suspended matter. The measurement technique involves light scattering. The instrument is the turbidimeter, consisting of a nephelometer, light source, and photodetector. The standard unit is the nephelometric turbidity unit (NTU). The correlation between TSS and NTU is often good for a specific wetland system, but care must be taken in the extrapolation from one site to another (Table 7.1). From these results, it may be concluded that the NTU–TSS relationships for FWS wetland effluents differ substantially from those for activated sludge effluents, and vary somewhat between natural systems.

Potential for Sampling Errors

It is sometimes virtually impossible to sample interior wetland waters for TSS because of the disturbance of sediments caused by sampling. Errors of one to two orders of magnitude can easily occur. This is the case in shallow zones of vegetated FWS wetlands. If the water is deeper than about 20 cm, accurate sampling is possible but not easy. Immersion of a sampler may cause disturbance of bed sediments, or the currents caused by water rushing into a sample bottle may disturb those sediments. Ideally, the sample should flow into the sample bottle at the local velocity of the water in the wetland. This is termed *isokinetic sampling*, and is necessary to prevent extraneous resuspension. It is often not possible to achieve undisturbed sampling for TSS, and therefore difficult to obtain proper flow-weighted or volume-weighted values of TSS at interior points. For this reason, much of the available TSS data from wetland treatment systems consists of input and output measurements in pipes and at structures.

This difficulty carries over to those chemical constituents which partition strongly to the solids, or form an integral part of them. Any interior water sample will likely contain an unrepresentative proportion of the locally agitatible, or transportable, sediments and particulates. Subsequent analysis for the total amount of a partitioned or contained substance will yield an inaccurately high value.

TABLE 7.1
Regressions between Total Suspended Solids and Turbidity for Wetlands, Forced through the Origin (TSS = 0, NTU = 0)

	NTU/TSS	R^2	TSS Range (mg/L)	Turbidity Range (NTU)	Number	Reference
Secondary effluent	0.37–0.50	—	—	—	—	Crites and Tchobanoglous (1998)
Secondary effluent	0.42–0.43	—	—	—	—	Metcalf and Eddy (1991)
Everglades	0.25	0.80	1–18	0.4–3.4	126	South Florida Water Management District, unpublished data
River water	0.83	0.77	0–145	0–125	64	Des Plaines River Project, unpublished data
River water	0.66	0.95	50–1,400	100–1,000	23	Harter and Mitsch (2003)
Agricultural runoff	0.75	0.52	—	—	1,013	Everglades Nutrient Removal Project, unpublished data
Submerged vegetation	0.74	0.93	0–215	0–150	>100	James *et al.* (2002)
Water hyacinths	1.39	0.54	4–18	6–21	12	Crites and Tchobanoglous (1998)
Oxidation pond	0.47	0.06	1–15	1–27	96	Gearheart *et al.* (1983)

Similar sampling problems exist for HSSF wetlands. Most of the solids present within a HSSF wetland bed are an accumulation of microbial biofilms, intercepted particulate matter, and plant-root networks. This accumulated material, collectively called a *biomat*, occurs either as material attached to the bed media and plant roots or as colloidal material within the media pores. Because the actual flow velocity, v (see Chapter 2), in an HSSF bed is very low, sampling events can induce localized flow velocities at the point of sample collection that are much higher than ambient flow velocities. This disturbs the *in situ* biomat and leads to sampling errors.

Introduction of sampling probes within the HSSF bed disturbs the bed matrix, shearing biomat off bed particles, which interferes with sample accuracy. As a result, samples taken within the HSSF bed are typically done using sample ports fabricated from perforated pipe (the same applies for VF wetlands). These sample ports are installed during construction and are a permanent feature of the HSSF wetland bed. Depending on the orientation of the perforated section of the pipe (horizontal or vertical), these sample ports will produce a sample that is width-averaged or depth-averaged over a localized portion of the HSSF wetland bed. A typical HSSF sample port assembly is shown in Figure 7.1; installation of the ports within an HSSF wetland is shown in Figure 7.2. However, the use of such pre-installed internal sampling ports does not guarantee that samples will be representative, because solids may still be selectively aspirated into the port. Difficulties in sampling lead to large variability for interior TSS samples. For instance, the coefficient of variation for TSS samples from the HSSF bed at Minoa, New York, was

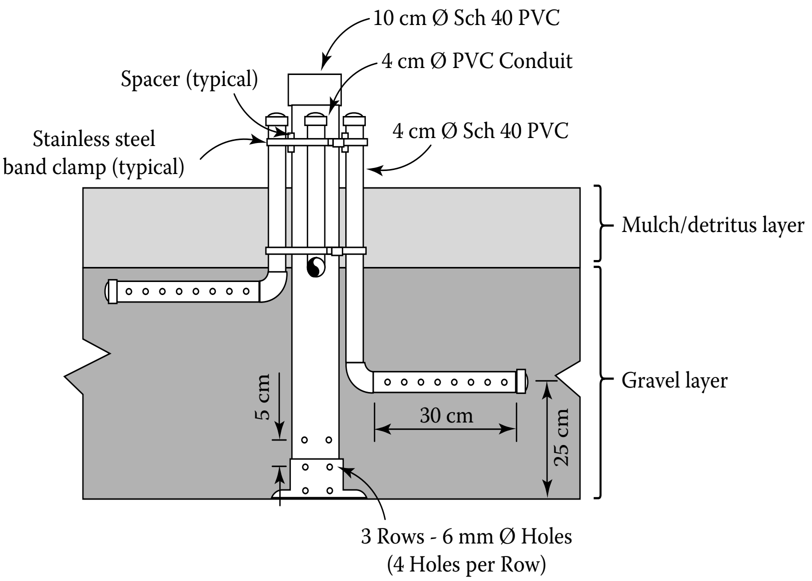


FIGURE 7.1 Example of a HSSF wetland sampling port. This particular assembly is designed to allow sample collection at three different bed depths and installation of a thermocouple at the base of the mulch layer.

FIGURE 7.2 Four-cell HSSF wetland at the University of Vermont. White pipes extending from the wetland beds are sampling ports.

72% (N = 534), with no apparent distance profiles. Similarly, the coefficient of variation was 145% (N = 215) in the Grand Lake, Minnesota, HSSF system.

As a consequence of these sampling difficulties, most of the samples collected in HSSF and VF wetlands consist of inlet and outlet samples, unless interior sampling ports were installed in the wetland at the time of construction. Because of the low flow velocities encountered in these systems, inlet and outlet works in contact with the water develop a biomat coating. Again, care must be taken not to disturb this biomat coating. If agitation of the water and sloughing of the biomat occurs, the sample will be contaminated and is no longer representative of the wastewater. As a result, high-energy devices such as dipping buckets and bailers should be avoided. The use of peristaltic pumps is one preferred sampling method, as the rate of sample withdrawal can be controlled, and the sampling tube can be carefully positioned to collect a representative sample. Small-diameter guide pipes are sometimes installed to facilitate placement of the sampler tubing away from side walls, tank bottoms, and other sources of sample contamination.

Solids Characterization

The suspended solids entering a treatment wetland may display widely varying characteristics, according to the source water involved. Domestic wastewaters at all pretreatment stages contain suspended materials that are primarily organic. Runoff waters, both urban and agricultural, may contain high proportions of mineral matter. Other source waters may involve highly specific characteristics, such as the colloidal materials that discharge from milking parlors. The two principal ways of describing solids are: the soil type and the size distribution.

Soil fractions are often also applied to suspended matter, especially for situations involving mostly mineral materials. These fractions are: organic, clay, silt, and sand. The VSS fraction of the solids is usually taken to be a measure of the organic fraction (Table 7.2), and the remaining nonvolatile suspended solids (NVSS) are assumed to be the mineral fraction of the overall TSS. For incoming waters derived from runoff from mineral soils, the fraction organic may be rather low. At the Des Plaines site, river water entering averaged 11–16%

TABLE 7.2
Organic Content of Various Source Waters Entering Treatment Wetlands

System	Influent Source	TSS Inlet (mg/L)	% NVSS
Houghton Lake, Michigan	Lagoon	25	56
Estevan, Saskatchewan	Lagoon	27	40
Des Plaines, Illinois	River	80	24
Tarrant, Texas	River	276	10
Tarrant, Texas	Sedimentation basin	37	20
Connell, Washington	Potato processing	350	94

Note: NVSS = non-volatile suspended solids

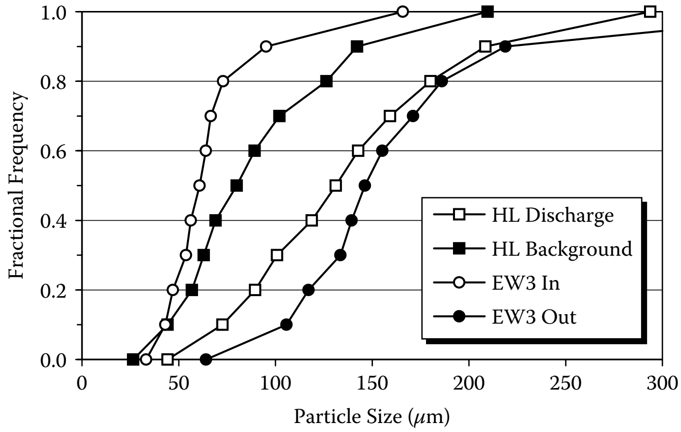


FIGURE 7.3 Particle size distributions for two FWS wetlands. At Des Plaines (EW3), the outlet particles are larger than those entering. At Houghton Lake (HL), the discharge area particles are larger than those in wetland background areas. (From unpublished data.)

organic, whereas water leaving the treatment wetlands averaged 16–26% organic. Harter and Mitsch (2003) reported 9% organic for both entering and leaving waters from the Olentangy River wetlands. However, the Houghton Lake natural peatland showed 77% organic, and after lagoon wastewater addition showed 56% organic (unpublished data). As an extreme example, the fraction VSS in a potato wastewater treatment wetland was 94% (unpublished data). Obviously, no generalizations may be made across the spectrum of treatment wetlands and source waters, but it should be noted that organic materials may be subject to decomposition after deposition.

Mineral constituents may be defined by size ranges (Lane, 1947; Brix, 1998; Braskerud, 2003):

Clay:	size < 2 μm
Silt:	2 μm < size < 60 μm
Sand:	60 μm < size < 2 mm
Gravel:	2 mm < size < 64 mm

These mineral particles have relatively high densities, $\rho_s \approx$ 2–2.5 g/cm^3, and the larger sizes settle readily. In contrast to organics, these materials accrete without decomposition.

Neither the particles entering the wetland nor those leaving are of a single size. Frequency distributions of particle sizes are always present (Figure 7.3). As a result, particle processing also becomes distributed, with large particles behaving differently from small.

7.2 PARTICULATE PROCESSES IN FWS WETLANDS

FWS wetlands process sediments and TSS in a number of ways (Figure 7.4). After the suspended material reaches the wetland, it joins large amounts of internally generated suspendable materials, and both are transported across the wetland. Sedimentation and trapping, and resuspension, occur en route, as does "generation" of suspended material by activities both above and below the water surface. For example, algal debris may form at one location and deposit downgradient in the wetland.

PARTICULATE SETTLING

Single Particles

The slow-moving waters in the FWS wetland environment often permit time for physical settling of TSS. The settling velocity of the incoming particulates, combined with the depth of the wetland, gives an estimate of the time and travel distance for those solids.

Solids sink in water due to the density difference between the particle and water. For single, isolated spherical particles, the terminal velocity is reached quickly:

$$w^2 = \frac{4}{3}\frac{gd}{C_D}\left(\frac{\rho_s - \rho}{\rho}\right) \tag{7.1}$$

where

d = particle diameter, m
C_D = drag coefficient, dimensionless
g = acceleration of gravity, m/s^2
w = terminal velocity, m/s
ρ = density of water, kg/m^3
ρ_s = density of solids, kg/m^3

In turn, the drag coefficient is a function of the particle Reynolds number:

$$C_D = \left(\frac{24}{Re_p}\right)\left(1 + 0.15\,Re_p^{0.687}\right) \tag{7.2}$$

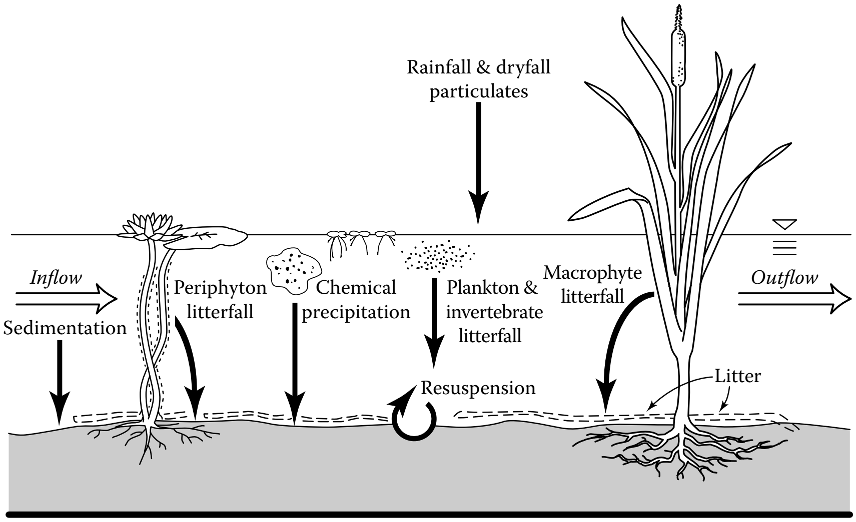


FIGURE 7.4 Processes affecting particulate matter removal and generation in FWS wetlands. (Adapted from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

where the particle Reynolds number is:

$$Re_p = \frac{d\rho w}{\mu} \tag{7.3}$$

where

Re_p = particle Reynolds number, dimensionless
d = particle diameter, m
ρ = density of water, kg/m^3
w = terminal velocity, m/s
μ = viscosity of water, kg/m·s (= 0.001 μ, in centipoise)

If all physical properties are known, Equations 7.1–7.3 combine to determine the settling velocity. This calculation is easily automated on a spreadsheet, with the results shown in Figure 7.5.

In the laminar flow region, $Re_p < 1.0$, the drag coefficient is inversely proportional to the particle Reynolds number, and the settling velocity of the particle is then calculable from *Stokes law*:

$$w = \frac{gd^2}{18\mu}\left(\rho_s - \rho\right) \tag{7.4}$$

where

d = particle diameter, m
g = acceleration of gravity, m/s^2
w = terminal velocity, m/s
ρ = density of water, kg/m^3
ρ_s = density of solids, kg/m^3
μ = viscosity of water, kg/m·s (= 0.001 μ, in centipoise)

In the wetland environment, neither the density nor the particle diameter is known, and the particles are not spheres or

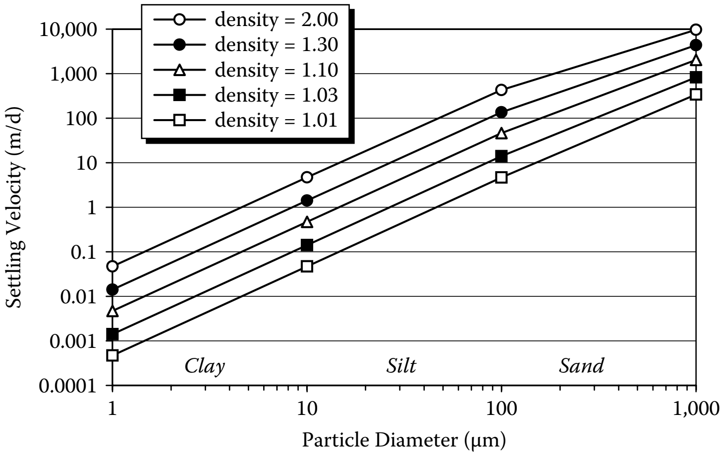


FIGURE 7.5 Settling velocity of spherical particles in water at 20°C, for different particle densities.

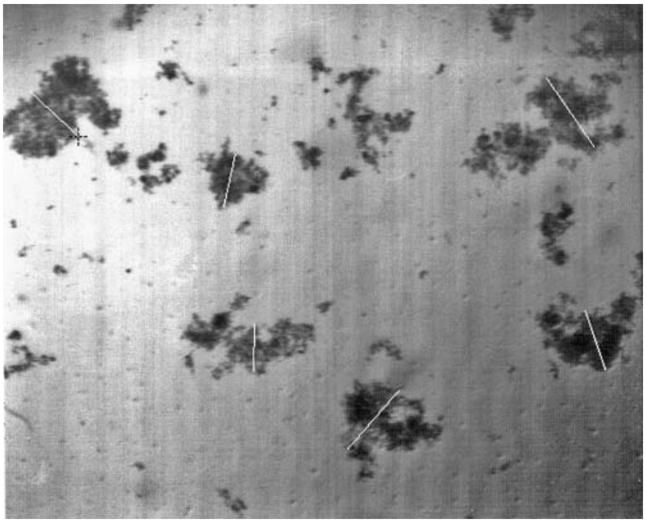


FIGURE 7.6 Photomicrograph of suspended particulate matter in the effluent from Des Plaines wetland EW3. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

discs (Figure 7.6). Although it is possible to correct for non-spherical shapes (Dietrich, 1982), there is not a convenient method for determination of the particle density. Further, particles may agglomerate to larger size, or be subject to interference from neighboring particles.

Settling of Mixtures

Settling of particulate matter may be described by a first-order model (Equation 7.4) for each size fraction. In general, settling velocities are proportional to the square of particle size, with variation including shape factors and particle density. Particle mass may be estimated to be roughly proportional to the cube of size. The time of fall of a particle through a vertical distance (h) is determined from its velocity:

$$t_{\text{fall}} = \frac{h}{w} \tag{7.5}$$

where

h = water depth, m
t_{fall} = time to fall, s
w = terminal velocity, m/s

If the water is moving through the wetland length (L) at velocity (u), the time of travel is:

$$t_{\text{travel}} = \frac{L}{u} \tag{7.6}$$

where

L = wetland length, m
t_{travel} = time to traverse wetland, s
u = superficial water (flow) velocity, m/s

Theoretically, all particles of a size corresponding to a given fall velocity will be removed by settling if the travel time exceeds the settling time from the top of the water:

when $$\frac{L}{u} > \frac{h}{w}$$

$$N_{\text{fall}} = \frac{Lw}{uh} > 1 \tag{7.7}$$

where

N_{fall} = particle falling number, dimensionless

These concepts have been applied to mixtures in shallow overland flow in grass (Deletic, 1999), and in wetlands (Li *et al*., 2007), with mean particle diameter used to determine the settling velocity (w). Values of N_{fall} were found to be above 10 for complete removal, reflecting the difficulty of settling of the small end of the particle size distribution (Figure 7.7).

These relations also allow the conversion of a size distribution to a settling velocity distribution, and ultimately to the size distribution remaining after some fixed settling time. Procedures for such calculations may be found in Crites and Tchobanoglous (1998); however, there is rarely sufficient information on particle properties available. Braskerud (2003) found considerable discrepancies when applying these procedures to mineral particles trapped in wetlands.

Column Studies

Settling rates may also be determined experimentally. Typically, a large diameter column of water is charged with a well-stirred suspension of particles, and the concentration measured at a sequence of times at a series of depths below the water surface. Vertical profiles of TSS exist in differing shapes, depending on flocculation and particle–particle interference. A number of analytical techniques may be applied to such data (Font, 1991). Only the mean water column concentration of

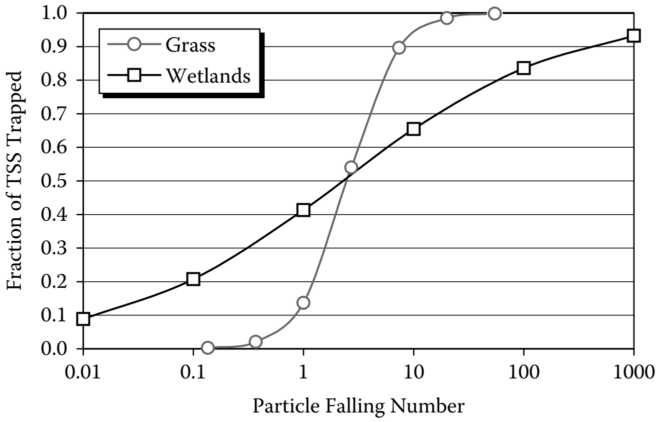


FIGURE 7.7 Removal of TSS in shallow overland flow in grass. The particle falling number is (Lw/uh), in which w is the terminal velocity of the mean particle diameter. Original data centered on a mean diameter of about 50 μm. (Data from Deletic (1999) *Water Science and Technology* 39(9): 129–136; and Li *et al*. (2007) *Journal of Hydrology* 338: 285–296.)

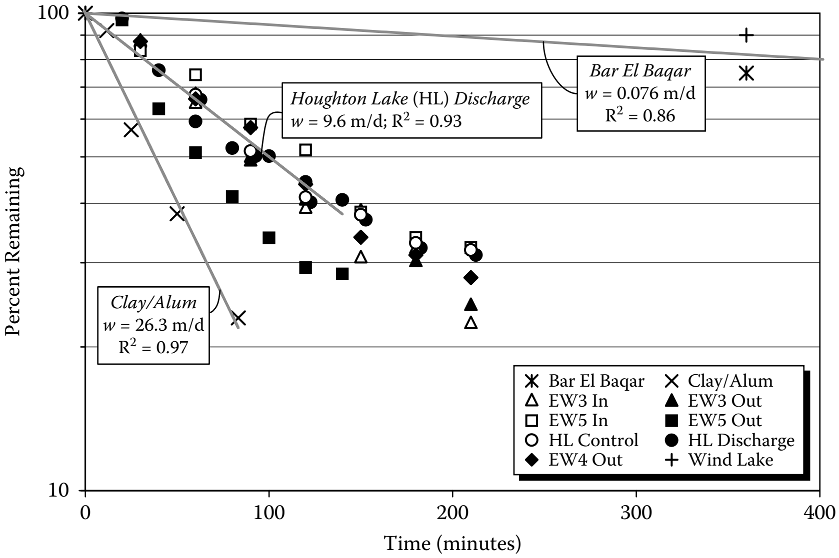


FIGURE 7.8 Examples of settling characteristics of TSS derived from wetlands and other natural contributing sources. The mean settling velocities range from 0.076 m/d for the Wind Lake wetland TSS, to 26.3 m/d for the clay alum mix. (Data for HL Control, HL Discharge, EW3 In, EW3 Out, EW4 Out, EW5 In, EW5 Out, and Wind Lake: authors' unpublished data; data for Clay/Alum: ASCE (1975) *Sedimentation Engineering*. Vanoni (Ed.), American Society of Civil Engineers (ASCE): New York; data for Bar El Baqar: PLA (1993) *1993 Field Program for the Egyptian Engineered Wetland*. Report prepared for the United Nations Development Programme, New York, P. Lane and Associates, Ltd. (PLA).) (Graph from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

TSS will be considered here. That concentration decreases as time progresses. Settling column data, for example, wetland waters and other sources, indicate an exponential decrease in concentration with time, and a time scale of a few hours for the majority of settling to occur (Figure 7.8). The settling velocities shown in Figure 7.8 range from $w = 0.076$ to 26.3 m/d. Interestingly, exponential decreases are found for the several sediments in Figure 7.8.

Caution must be used in those applications where colloidal materials may be present in the inflow, because these materials are stable or very slow to settle. Very fine clay suspensions and some milk processing wastewaters fall into this category. The settling velocity for planktonic solids was found to be on the order of $w = 0.076$ m/d for the Wind Lake, Wisconsin, wetland, which was dominated by algae.

Column settling data provide estimates of the removal time for TSS in the absence of dense vegetation. Confirmation of field applicability was found for wetland EW3 at Des Plaines in 1991. The inlet zone was essentially unvegetated, and the water velocity was on the order of 30 m/d. Settling column data (Figure 7.8) suggested that solids should essentially be gone in eight hours, or after a travel distance of about ten meters. Transect information confirmed this estimate.

"Filtration" versus Interception

Conventional wisdom has it that the presence of dense wetland vegetation causes settling to be augmented by filtration. This is often not true in the usual sense of the term filtration. It is trapping of sediments in the litter layer that prevents resuspension, and thus enhances the net apparent suspended sediment removal. Macrophytes and their litter form a nonhomogeneous "fiber bed" in the wetland context. The void fraction in the stems and litter is quite high; straining and sieving are thus not typically the dominant mechanisms. Submerged biomass additionally traps sediment in sheltered microzones, thereby lessening the potential for resuspension. Confirmation of sedimentation as the principal mechanism was provided in the laboratory studies of Schmid *et al.* (2005).

However, there are wetland circumstances in which the dominant mechanism is particles striking immersed objects and sticking. The three principal mechanisms of fiber-bed filtration are well known and documented in handbooks (see, e.g., Perry *et al.*, 1982; Metcalf and Eddy, 1991):

1. Inertial deposition or impaction—particles moving fast enough that they crash head-on into plant stems rather than being swept around by the water currents.
2. Diffusional deposition—random processes at either microscale (Brownian motion) or macroscale (bioturbation) which move a particle to an immersed surface.
3. Flow-line interception—particles moving with the water and avoiding head-on collisions, but passing close enough to graze the stem and its biofilm, and sticking.

The efficiencies of collection for these mechanisms depend on the water velocity, particle properties, and water properties, as well as the character of submerged surfaces. A typical wetland "fiber" is a bulrush stem of about 1 cm diameter.

A typical particle might be on the order of 1–100 μm. A typical water velocity is on the order of 10–100 m/d. Under these conditions, the collection efficiencies of Mechanisms 1 and 2 are predicted to be vanishingly small. There is evidence that Mechanism 3 is operative and significant. Lloyd (1997) examined the submerged surfaces of bulrushes (*Schoenoplectus* (*Scirpus*) *validus*) and found particles as small as 0.5–2.5 μm sticking to biofilms (Breen and Lawrence, 1998). Saiers *et al.* (2003) studied the movement of very small (0.3 μm), unsettleable particles of TiO_2 in the Florida Everglades. They concluded that 29% of the particle impacts on periphyton-coated stems resulted in sticking in a plant (*Eleocharis* spp.) density of 1,150 per m². These stems were only 0.2 cm in diameter, resulting in 99% porosity. Saiers *et al.* (2003) defined a first-order rate constant for removal by sticking, which on an areal basis is:

$$k = \frac{\eta u h^2}{\left[\dfrac{1}{n} - \dfrac{\pi d^2}{4}\right]} \tag{7.8}$$

where
- d = stem diameter, m
- h = water depth, m
- k = areal removal rate constant, m/hr
- n = stem density, #/m²
- u = water velocity, m/hr
- η = sticking efficiency, dimensionless

Resuspension

Settled particles may not "stay put" for a number of reasons. Hydrodynamic shear forces may tear particles loose from the sediment bed, which is a dominant mechanism in streams and rivers. However, wetlands provide an environment in which other processes may occur as well. Wind and wave action are major drivers of resuspension in lakes, and may also be operative in open water areas of FWS wetlands. Additionally, biological activity may result in the movement of particles from the sediments to overlying water.

Unvegetated Surfaces

Much is known about the resuspension of particulates from flat surfaces (ASCE, 1975). Most interpretations are made in terms of the force per unit area (shear stress) required to tear a particle loose from the sediment surface. The concepts involve purely physical forces and apply most readily to mineral substrates and river systems. Most theoretical results are for planar sediment bed bottoms with no extraneous objects. Vegetated wetland bottoms do not fit these conditions.

In the treatment wetland environment, physical resuspension (due to high flow velocities) is not a dominant process. Water velocities are usually too low to dislodge a settled particle from either the bottom or a position on submerged vegetation. However, in design, it may be necessary to avoid wetland aspect ratios that produce excessively high linear velocities. The potential for erosive velocities exists for highly loaded wetlands with high length-to-width ratios. Estimation of the velocity required to foster resuspension may be based on the settling characteristics of the solids and the frictional characteristics of the wetland, combined with known correlations of the critical shear stress for particle dislodgment (ASCE, 1975). Modifications are needed for the case of laminar flow, which is the general case for wetlands (Mantz, 1977; Yalin and Karahan, 1979).

Velocities that cause erosion in open channels are high compared to wetlands. For instance, French (1985) lists recommended maximum (nonscouring) velocities for 14 canal materials in the range $0.46 < u < 1.83$ m/s. Such considerations resulted in a maximum canal velocity design constraint of 0.76 m/s for Everglades protection wetlands conveyance canals (Burns and McDonnell, 1996). In anticipation of more erodable particulates inside the wetlands, wetland velocities were limited to no more than 0.03 m/s (2,600 m/d). These large wetlands had lengths up to 2,500 m, which therefore created a design detention minimum of one day. The annual average design detention time was 30 days. No erosion has been noted in this project or its companions of comparable size and detention.

Effects of Vegetation

It is known that vegetation increases the retention of particulates in both lake and stream environments. For instance, Horppila and Nurminen (2003) found that beds of submerged plant species—butter cup: *Ranunculus circinatus*; coontail: *Ceratophyllum demersum*; and pond weed: *Potamogeton obtusifolius*—in a lake environment effectively prevented resuspension, which they attributed to a reduction in wind and wave action. Horvath (2004) studied the effect of macrophytes—rushes: *Juncus* spp.; bur-reed: *Sparganium* spp.; forget-me-not: *Myosotis* spp.—on retention of particulate matter in a small stream, and found enhanced trapping in proportion to biomass.

It is logical that these same effects are prevalent in treatment wetlands. Dieter (1990) found about a threefold reduction in resuspension from open water to vegetated areas in a prairie pothole wetland. Hosokawa and Horie (1992) demonstrated enhanced removal in both laboratory channels with dowels and in field flumes in a reed bed (*Phragmites australis*). In fully vegetated wetlands, the litter and root mats provide excellent stabilization of the wetland soils and sediments. This limits, but does not eliminate, resuspension.

The Floc Layer

Some treatment wetlands, such as those used for low-level nutrient removal, develop very flocculent sediment beds. These sediments are positioned on top of the consolidated soils, and may be interwoven with plant detritus. Bulk densities of such floc layers may range downward to 0.03–0.05 g/cm³ of dry matter (James *et al.*, 2001; Coveney *et al.*, 2002). Depths of these loose and unconsolidated materials have been found to exceed 30 cm in some situations (Table 7.3).

TABLE 7.3
Floc Thicknesses and Bulk Densities for the Everglades Nutrient Removal Project (ENRP), Lake Apopka, Florida Project, and the Sacramento California Demonstration Wetlands Project

		Thickness (cm)			Bulk Density (g/mL)		
Site	**Years**	**Mean**	**SE**	**N**	**Mean**	**SE**	**N**
Sacramento 4	2.6	17.2	1.4	8	0.068	0.015	12
Sacramento 4	2.6	11.3	1.0	8	0.069	0.017	16
ENRP 1	9.0	19.7	1.4	30	0.076	0.006	30
ENRP 2	9.0	18.2	1.4	26	0.099	0.007	26
ENRP 3	9.0	18.9	1.8	22	0.072	0.008	22
ENRP 4	9.0	16.7	1.4	10	0.092	0.012	10
Apopka	2.4	33	—	48	0.051	—	48

Source: Data from Nolte and Associates (1997) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project*. 1996 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates: Sacramento, California; Coveney *et al.* (2002) *Ecological Engineering* 19(2): 141–159; and South Florida Water Management District, unpublished data.

Despite low bulk density, the amount of floc dry matter is substantial. For instance, the Sacramento data in Table 7.3 convert to about 9,700 g/m^2 of dry matter present as the floc.

The origins of floc are not well understood, but it has been found to occur in both macrophyte-dominated (Sacramento) and SAV-dominated (ENRP Cell 4) wetlands. It likely contains a significant microbial detrital component, as well as algal and macrophyte detritus. Floc also occurs in the ultra-low nutrient, unimpacted Everglades (Gaiser *et al*., 2005), where it is presumably the result of an active periphyton biological cycle.

There is not an accepted common terminology for the floc. Nolte (1997) called it the "*A* layer," and described it as follows:

> The *A* layer consists of a slurry of dark, decomposing, loosely structured detrital material that pours out when the sampler is tipped. The material in the *A* layer has settled to the bottom, but has not been integrated into the matrix of the basin floor.

This material is not subject to transport under most ambient conditions, but is very mobile if disturbed. For example, disturbance resuspension tests were conducted at the Houghton Lake treatment wetland. A bottomless sharp-edged cylinder was twisted down into the soil, and the interior biomass (live, dead, litter) was removed. The remaining, isolated water was gently agitated, and then sampled for solids content. The mobile material averaged 880 ± 100 g/m^2 (mean ± SE).

Other Resuspension Mechanisms

The wetland environment provides an opportunity for three other mechanisms of resuspension: wind-driven turbulence, bioturbation, and gas lift. In open water areas, wind-driven currents cause surface flow in the wind direction and return flows along the bottom in the opposite direction. These recirculation velocities can far exceed the net velocity from inlet to outlet. For wetlands with large open water zones, waves add to the overall process of resuspension. Lake studies suggest both processes are wind-dependent. For instance, Malmaeus and Hakanson (2003) suggest resuspension is proportional to the square of the wind speed. Additionally, fetch and water depth are controlling factors.

Animals of all types and sizes can cause resuspension to occur. Feeding carp (Kadlec and Hey, 1994) and nesting shad (APAI, 1995) have been observed to cause problems. The carp rooted in the sediments for food, and thus resuspended large amounts of sediments. Control was by drawdown and freezing. The shad fanned nests on the wetland bottom, and resuspended sediments. Control was by drawdown and avian predation. Beaver activity can cause stirring, often at the outlet of the wetland, in conjunction with attempts to dam the outlet. Human sampling activities in the interior of treatment wetlands may also result in locally-elevated concentrations of suspended solids. For instance, the passage of a drifting boat can cause extreme resuspension (Figure 7.9).

Gas lift occurs when bubbles of gas become trapped in or attached to particulate matter. Wetland sediments are often of near neutral buoyancy; so a small amount of trapped gas can cause "sinkers" to become "floaters." There are several gas-generating reactions in a wetland environment. Most important are photosynthetic production of oxygen by algae and production of methane in anaerobic zones.

Chemical Precipitates

Several chemical reactions can produce particulate matter within wetlands under the proper circumstances. Some of the more important are the oxyhydroxides of iron, calcium carbonate under aerobic conditions, and divalent metal sulfides under anaerobic conditions. As conditions of chemical composition, pH, and redox change in the wetland, these and other compounds may undergo dissolution and be removed from the sediment bed.

FIGURE 7.9 Passage of a drifting boat can stir up a cloud of floc. This site is in the interior of the A.R. Marshall Loxahatchee National Wildlife Refuge. The water was about 45 cm deep, and the vegetation was sparse.

Iron Flocs. The iron oxyhydroxides are typically flocs, with the possibility of coprecipitates. They may form under conditions of elevated dissolved ferric iron and oxygen-rich water. The processes may be represented as (Younger *et al.*, 2002)

$$Fe^{2+} + \frac{1}{4}O_2 + H^+ \rightarrow Fe^{3+} + \frac{1}{2}H_2O \tag{7.9}$$

$$Fe^{3+} + 2H_2O \rightarrow FeOOH_{(sus)} + 3H^+ \tag{7.10}$$

$$FeOOH_{(sus)} \rightarrow FeOOH_{(sed)} \tag{7.11}$$

These precipitates are characterized by an unmistakable blood-red color (Figure 7.10). As indicated by the chemistry, formation is inhibited by low pH and by low dissolved oxygen. Formation may be abiotic, or mediated by microorganisms such as *Thiobacillus ferrooxidans*. However, at pH > 9, the rate of the abiotic reaction is so fast that formation is controlled by the rate of oxygen supply (Younger *et al.*, 2002). In the pH range 6 < pH < 8 that generally typifies treatment wetlands, rates are slow enough to be a design consideration. This set of reactions forms the basis for phosphorus removal by addition of ferric chloride to wastewaters, and the accompanying co-precipitation of the phosphorus. Consequently, the subsequent fate of these solids in polishing treatment wetlands is of considerable interest.

Aluminum Flocs. The aluminum oxyhydroxides are also typically flocs, with the possibility of co-precipitates. They may form under circumneutral pH conditions, and do not require oxygen. The processes may be represented as (Sobolewski, 1999):

$$Al^{3+} + H_2O \rightarrow Al(OH)_3 \downarrow + 3H^+ \tag{7.12}$$

FIGURE 7.10 (A color version of this figure follows page 550) Venting groundwater at this Wellsville, New York, site contains iron, which oxidizes upon contact with air.

These precipitates are characterized by their formation of a "pin floc" material that does not readily settle in FWS wetlands (Bachand *et al.*, 1999). This set of reactions also forms the basis for phosphorus removal by addition of alum to wastewaters, and the accompanying co-precipitation of the phosphorus. Consequently, the subsequent fate of these solids in polishing treatment wetlands is of considerable interest.

Calcium Carbonate. Calcium carbonates may be formed in wetlands, under conditions of elevated pH and dissolved calcium. The operative chemistry may be summarized as

$$Ca^{2+} + HCO_3^- + H_2O \rightarrow CaCO_3 \downarrow + H^+ \quad (7.13)$$

This reaction may occur abiotically, but perhaps more importantly it may be mediated by algae. Algal activity can drive up pH, and create conditions that foster creation of calcium-rich solids (Vymazal, 1995). Indeed, this process has contributed to the formation of marl prairies as a form of natural wetlands. New sediments in Everglades protection treatment wetlands contain a significant fraction of calcium compounds (Dierberg *et al.*, 2002).

Metal Sulfides. Many metals form very insoluble sulfides, including mercury, lead, cadmium, and zinc, as further discussed in Chapter 11. These precipitates are important in the processes of metal removal in wetlands, and follow the general chemistry (Sobolewski, 1999):

$$SO_4^{2-} + 2CH_2O \rightarrow HS^- + H^+ + 2HCO_3^- \quad (7.14)$$

$$M^{2+} + HS^- \rightarrow MS + H^+ \quad (7.15)$$

However, for many treatment wetland applications, metals are present at only very low concentrations. Consequently, the formation of insoluble sulfides does not usually create measurable additions to the sediments of the wetlands.

Biological Sediment Generation

Wetlands produce sediments via processes of death, litter fall, and litter attrition. This occurs for biota at a number of different size scales, ranging from macrophytes on down to bacteria. Algal productivity can be a major generator of suspended solids. A second set of processes adds pollen and seeds to the water. The TSS produced is organic in character, resulting in a high carbon content and a high proportion of VSS. The chlorophyll and pheophytin (dead chlorophyll) content is high if the algal pathway is dominant.

Some TSS originates from leaf and stem litter. For instance, annual leaf litterfall in a natural sedge-shrub peatland was found to be 60–70 g/m^2 (Chamie, 1976). Some part of this material contributes to TSS, either via direct attrition, or via microbial decomposition.

The generation of sedimentary material is a very important internal process in nutrient-rich treatment wetlands. The generous supply of nutrients assures a large production of a wide variety of transportable organisms and associated dead organic material. Such wetlands are characterized by high water chlorophyll content and high sediment accumulation. Bacterial and algal growth is promoted, and decomposition products form a new pool of suspendable material. A host of wetland invertebrates, such as *Daphnia* and waterboatman (*Corixidae*), also die and contribute to the sediments, and they may be present in pumped lagoon water.

These processes are virtually impossible to predict and quantify. But it is important to recognize that they exist, because they contribute to a background level of TSS in a wetland.

Accretion

Trapped TSS, plus material generated within the wetland, will accrete as either movable sediment or the consolidated immovable new soil produced from the sediments. Not all of the dead plant material undergoes decomposition. Some small portions of both aboveground and belowground necromass resist decay, although these are typically shredded by microbial and other invertebrate processes. Underground processes form nonsuspendable accretions, some part of which is stable and does not fully decompose. The origins of new sediments may be from remnant macrophyte stem and leaf debris, remnants of dead roots and rhizomes, and from indecomposable fractions of dead microflora and microfauna (algae, fungi, invertebrates, bacteria).

Measurement of Accretion

The processes above combine to determine the amount of sediment at various locations within the wetland as a function of time and the TSS concentration in the wetland effluent. Cup collectors may be placed on the wetland bottom (Jordan and Valiela, 1983; Fennessy *et al.*, 1992; Braskerud, 2001a); these typically intercept the downward vertical flux of sediment but prevent shear-induced resuspension. Plate collectors may be placed on the wetland bottom, followed by sediment harvest above that horizon at a later time (Kozerski and Leuschner, 1999; Braskerud, 2001a). Alternatively, neutral density particulate material may be laid down in a layer, and retrieved by coring and sectioning (Harter and Mitsch, 2003). Another technique involves the elevation of a blunt-footed rod, which is lowered to the sediment surface. A reference rod, driven deep into stable soils, provides the local datum (Reeder, 1990). Other quantitative studies have relied upon atmospheric deposition markers such as radioactive cesium (^{137}Cs) or radioactive lead (^{210}Pb) (Kadlec and Robbins, 1984; Craft and Richardson, 1993; Robbins *et al.*, 2004). These techniques require several years of continued deposition for maximum accuracy.

Cup collectors typically yield much more sediment than plate collectors. For instance, Schulz *et al.* (2003b) found 30 ± 3 g/m^2·d collected in cups in a riverine bed of *Sagittaria sagittifolia*, compared to 8 ± 2 g/m^2·d collected on plates. This is presumably due to the prevention of resuspension in cups, whether it be due to fluid shear or to bioturbation. For mineral sediments, the difference between cups and plates is less, probably because of the lesser importance of resuspension of heavier particles (Braskerud, 2001a).

TABLE 7.4
Accretion Rates in FWS Wetlands

Location	Wetland	Reference	Method	Water NH_3-N (typical) (mg/L)	Accretion (cm/yr)
Louisiana	Salt marsh	DeLaune *et al.* (1978)	^{137}Cs	Low	1.1–1.35
Louisiana	Forested	Conner and Day (1991)	Feldspar	Low	0.84
Louisiana	Forested	Rybczyk *et al.* (2002)	Feldspar	0.05	0.14
Xianghai, China	Open marsh	Wang *et al.* (2004)	^{137}Cs + ^{210}Pb	Low	0.35
Xianghai, China	Isolated marsh	Wang *et al.* (2004)	^{137}Cs + ^{210}Pb	Low	0.65
Michigan	Marsh	Kadlec and Robbins (1984)	^{210}Pb	0.1	0.2
Norway Farm Runoff	Marsh CW	Braskerud (2001b)	Plate	0.16	2
Norway Farm Runoff	Marsh CW	Braskerud (2001b)	Plate	0.37	4
Everglades WCA2A	Marsh	Reddy *et al.* (1993)	^{137}Cs	0.3	0.5
Everglades WCA2A	Marsh	Craft and Richardson (1993b)	^{137}Cs	0.3	0.4
Everglades WCA3	Marsh	Craft and Richardson (1993b)	^{137}Cs	0.1	0.3
Everglades	Marsh	Robbins *et al.* (1999)	^{210}Pb	0.3	0.5
Everglades	Marsh	Chimney (unpublished data)	Feldspar	0.1	0.85
Sacramento, California	Marsh CW	Nolte and Associates (1998b)	Visual	16	1.5
Houghton Lake, Michigan	Marsh NTW	Kadlec (unpublished data)	Resurvey	10	1.0
Chiricahueto Runoff, Mexico	Marsh	Soto-Jimenez *et al.* (2003)	^{210}Pb	14	1.0
Louisiana	Forested NTW	Rybczyk *et al.* (2002)	Feldspar	15	1.14

Note: CW = constructed wetland; NTW = natural treatment wetland.

Amount and Distribution of Accretion

Accretions measured in various wetlands vary from a few millimeters per year to over a centimeter per year (Table 7.4). These accumulated solids represent the potential for filling of a constructed wetland. It is an easy calculation to allocate the removed TSS to the buildup of new solids in the FWS wetland. For municipal wastewater polishing, typical operations lead to an accumulation of 1–2 mm/yr of new solids (50 mg/L removed at q = 5.5 cm/d at a bulk density of 0.5 g/cm^3 yields 2.0 mm/yr). But that material is augmented by internally generated solids and decreased by decomposition of the organic portion of sediments and soils. The net increase may total up to 10 mm/yr in a highly eutrophic marsh (Table 7.4). Even more accumulation can result from the trapping of mineral solids from urban or agricultural runoff.

For high amounts of sediment trapping compared to generation and resuspension, buildup typically occurs preferentially in the inlet section of the wetland. Therefore, a "delta" of accreted sediments builds in the inlet region of the wetland. For example, food processing wastewaters can contain very high TSS concentrations, which in turn can fill a treatment wetland with solids. Van Oostrom (1995) reported that one third of the volume of a floating *Glyceria* mat wetland was filled after 20 months of operation (Figure 7.11). The wastewater was

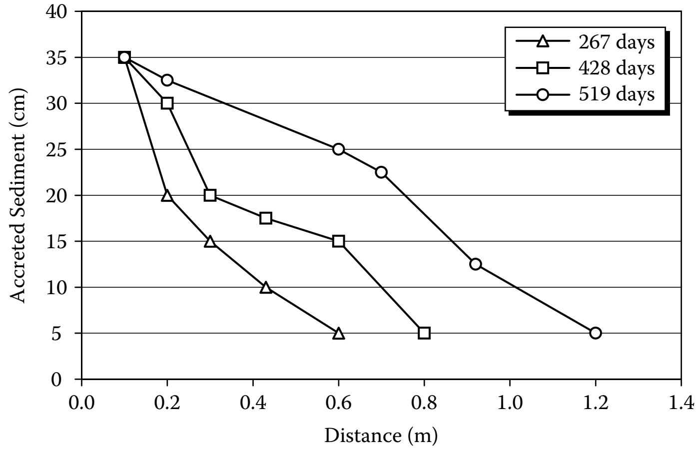


FIGURE 7.11 The sediment "delta" developed in a small treatment wetland mesocosm. (Data from van Oostrom (1995) *Water Science and Technology* 32(3): 137–148.) (Graph from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

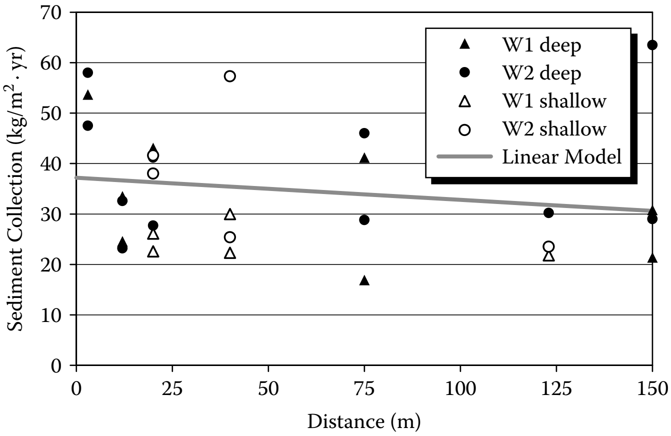


FIGURE 7.12 Spatial distribution of plate sediment collection rates along the flow direction of a constructed marsh treating river water. (Data from Harter and Mitsch (2003) *Journal of Environmental Quality* 32(4): 325–334.)

a nitrified meat processing effluent, with incoming TSS of 269 mg/L, and the removal rate was 5,300 g/m²·yr. Accreted sediments totaled 40% of the removed solids, 2,100 g/m²·yr, and these were concentrated near the inlet end of the wetland. The density of the solids was very low, around 0.03 g/cm³.

In contrast, lighter loadings and open water areas may foster the redistribution of suspendable material. For instance, Brueske and Barrett (1994) found a "delta" in a highly loaded wetland (around 3.6 g/m²·d TSS), but little or no "delta" for a lower loading (around 0.8 g/m²·d TSS). Both Harter and Mitsch (2003) and Brueske and Barrett (1994) found greater sediment accretion in open water areas, which may have been attributable to most of the flow traveling through such areas, or to bioturbation (Figure 7.12). In contrast, Benoy and Kalff (1999) found a linear relation between sediment accumulation and biomass for submerged species *Myriophyllum spicatum*, *Potamogeton* spp., *Ceratophyllum demersum*, and *Elodea canadensis* beds in Lake Memphremagog between Québec and Vermont. It is apparent that the processes involved in sediment accumulation in wetlands are too complicated to permit generalities.

In the long run, solids accretion may raise the elevation of the wetland bottom, and thus impact system hydraulics and treatment. U.S. EPA (2000a) suggests that accretion in municipal wastewater treatment wetlands results from both external and internal sources, which is conceptually correct. However, the U.S. EPA (2000a) estimate of accretion from external solids, 2–4 cm/yr, is based upon lagoon accumulation rates, and is excessively high. For example, the removal of 30 mg/L of TSS at a hydraulic loading rate of 10 cm/d results in solids storage of 1,095 g/m²·yr. At a density of 0.2 g/cm³, this gives 0.55 cm/yr if there is no decomposition. However, municipal TSS is about half mineral, and half-decomposable solids (VSS, see Table 7.2), and hence long-term external accretion would be about 0.27 cm/yr. U.S. EPA (2000a) estimates internal accretion as the annual deposition of macrophyte detritus to be 2.4 cm/yr. However, that material too is subject to decomposition, leaving an estimated residual long-term buildup of 20% of the input, or 0.48 cm/yr. In sum, the accretion in this example would be 0.75 cm/yr. This is consistent with the measured accretions in Table 7.4, for municipal systems. However, as the mineral content and loadings of TSS increase, so do accretions. Highly loaded wetlands treating mineral solids have been observed to accrete 2–8 cm/yr (Braskerud, 2001a).

Accretion is typically spatially nonuniform, due to gradients in deposition and productivity. This has been found to be true even in wetlands of very low nutrient status (Reddy *et al*., 1993). Inlet zones may therefore be subject to solids accumulations that are double the wetland average. However, some wetlands appear to redistribute solids fairly evenly from inlet to outlet.

To the authors' knowledge, only one municipal wastewater polishing FWS wetland has been serviced for solids removal, the Orlando, Florida Easterly Wetland inlet cells (White *et al*., 2004). The one removal of accumulations restored good hydraulic patterns, and restored original water quality performance.

It was suspected that uneven accumulations of new sediments were affecting flow patterns, and reducing efficiency (Sees, 2005). The inlet 9% of the wetland was excavated 45 cm, after 15 years of operation. This overexcavation restored more than the original freeboard, and resulted in a great improvement in hydraulic efficiency, from 34% to 74% (see Chapter 2). Two of the oldest facilities, Vermontville, Michigan (32 years, constructed), and Houghton Lake, Michigan (30 years, natural), have experienced accretions in the range of Table 7.4, but this has not jeopardized containment or operability. However, the Tucson, Arizona, Sweetwater wetland inlet cells have required solids removal after just a few years, because of the high suspended solids inlet water (see Figure 7.13).

FIGURE 7.13 Excessive TSS can fill the inlet deep zone to a treatment wetland, as happened at the Tucson, Arizona, sweetwater wetland. Note the bird tracks that highlight the complete filling of the deep zone with relatively high density solids. Incoming waters had high TSS from filter backwashes at the secondary treatment plant that provided the source water.

7.3 TSS REMOVAL IN FWS WETLANDS

As for most treatment wetland water quality parameters, the utilization of input and output data to compute percent removals is an inadequate representation of the processes which lead to those removals. This is particularly true for the removal of TSS.

Internal Cycling: Mass Balances

Models of sediment transport have been developed and verified for estuaries (Hayter and Mehta, 1986; Nakata, 1989, for example). These are 2- and 3-D models that allow for dispersion, settling, and resuspension; and generation is not usually an important term. These models may be adapted to the wetland situation. In the short term, there are significant fluctuations in TSS storage within the water column in response to the variations in settling, resuspension, and generation. Childers and Day (1990) state: "Our results affirm the variability of short-term sediment transport and depositional processes.…" Over a long period, however, changes in water column storage are negligible compared to other inputs and outputs. The water column TSS mass balance then assumes the character of a steady state model. There is an accompanying sediment bed balance, in which the change in storage is the dominant feature. The long-term, time-average profiles calculated from the vertically averaged mass balances for TSS in a linear flow wetland are (see Figure 7.14):

$$uh\frac{\partial C}{\partial x} = G + R - S \tag{7.16}$$

$$\frac{\partial(B+P)}{\partial t} = S - R - D \tag{7.17}$$

where

B = transportable solids bed, g/m^2
C = concentration, g/m^3 = mg/L
D = decomposition rate of transportable solids, g/m^2·d
G = generation rate, g/m^2·d
h = water depth, m
P = permanent soils and sediments, g/m^2
R = resuspension rate, g/m^2·d
S = settling rate, g/m^2·d
t = time, d
u = superficial water velocity, m/d
x = distance, m

In general, the settling rate may be written as:

$$S = w\,C \tag{7.18}$$

where

w = solids settling velocity, m/d

It is possible to derive two very useful results from these mass balances.

The w-C^* Model

First, in a spatially uniform wetland, as may occur after inlet settling effects no longer prevail, there will be no concentration gradient, and:

$$wC^* = G + R \tag{7.19}$$

where

C^* = uniform downgradient concentration, g/m^3 = mg/L

Second, if it is assumed that generation and resuspension are constant over the entire wetland, Equation 7.16 may then be written, for the plug flow assumption, as

$$u\,h\frac{dC}{dx} = w(C^* - C) \tag{7.20}$$

Integration from inlet to outlet then gives

$$\frac{(C_o - C^*)}{(C_i - C^*)} = \exp\left(-\frac{wL}{uh}\right) = \exp\left(-\frac{w\tau}{h}\right) \tag{7.21}$$

where

C_o = concentration, g/m³ = mg/L
C_i = concentration, g/m³ = mg/L
L = wetland length, m
τ = nominal detention time, d

The tanks-in-series (TIS) equivalent is (see Chapter 6):

$$\frac{(C_o - C^*)}{(C_i - C^*)} = \left(1 + \frac{wL}{Nuh}\right)^{-N} = \left(1 + \frac{w\tau}{Nh}\right)^{-N} \tag{7.22}$$

where

N = number of TIS

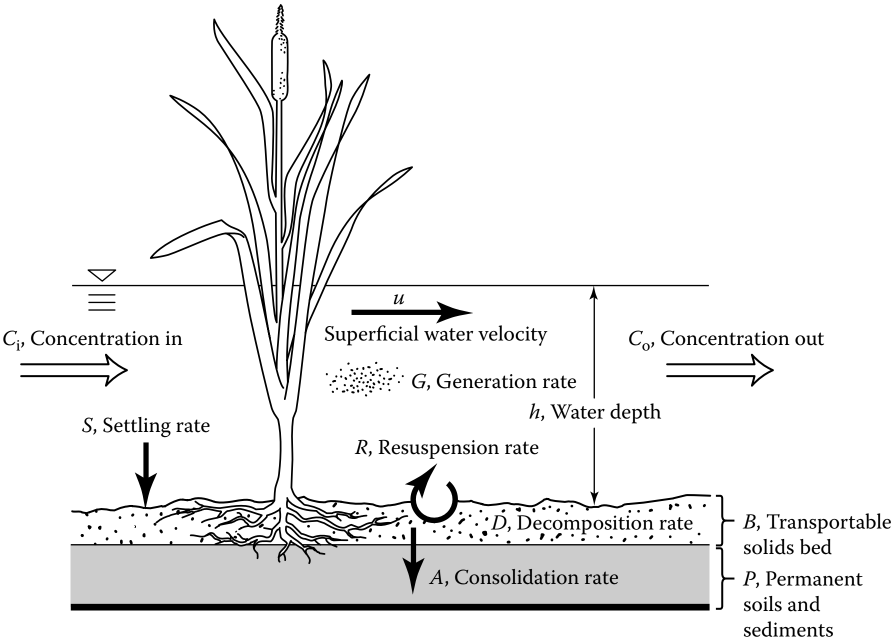


FIGURE 7.14 Framework for mass balances on suspendable materials in the wetland environment. (Adapted from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

Equation 7.21 contains a subtle message that bears on the removal of nearly all pollutants in wetlands, not just TSS. The right-hand numerator contains the settling velocity times the wetland length. An increase in either will cause a faster approach to C^*. The denominator contains the water velocity times the depth (uh). An increase in either of those will cause a slower approach to C^*. The detention time does not appear directly in this simplified mechanistic model, and the reason is easy to understand. If the water depth is doubled, for the same incoming volumetric flow rate and wetland area, the detention time will be doubled. But the particles do not fall any faster and now have twice as far to travel to the bottom. The extra detention time is used up by a greater vertical travel time. On the other hand, doubling the area of the wetland, all else being equal, will also double the detention time. The vertical settling distance is not increased, and the extra time causes greater removal.

A detailed gradient study to provide calibration of the k-C^* model (as discussed in Chapter 6) was done at the Hallam Valley wetlands in Melbourne, Australia (Wong *et al*., 2006). Exceedingly high water flows (nominal HRT < three hours) were required to detail the rapid decrease of TSS. Model fits were excellent, with w-values in the range of 16–21 m/d, for both vegetated and unvegetated channels. However, the C^*-value for the unvegetated channel was about double that for that for the vegetated channel (60 versus 33 mg/L). This is consistent with resuspension being greater in the open channel (Equation 7.19). The rates of TSS removal in other continuous flow through wetlands are not quite exponential (Figure 7.15) The rapid initial declines in concentration prevail for only a brief time of travel, after which declines follow a slower pace. (The Hallam Valley study did not contain a long portion of wetland that could display such a slow decline.)

Thus it is clear that the TSS leaving an FWS treatment wetland of moderate to long detention is more reflective of generation and resuspension than of unsettled incoming solids. Therefore, for nearly all FWS data sets, the parameter w cannot be determined accurately.

Internal Cycling

The second feature of the mass balances is the ability to measure individual components of solids processing, and to combine them to infer other results. Data from the Des Plaines may be used in this way. Wetland EW3 was heavily loaded when the pump was operating and contained relatively sparse emergent vegetation. Independent measurements were made in settling columns, yielding $w = 9.7$ m/d. Measurements of R were made utilizing sediment cups plus input and output data, which gave $R = 46.0$ g/m^2·d. Estimates of $G = 1.6$ g/m^2·d (WRI, 1992). Accordingly, from Equation 7.19, the expected value of $C^* = 4.9$ g/m^3. Thus both C^* and w were estimated independently from the transect data for TSS. The predicted drop in TSS agreed quite well with the measurements.

This same data gives allows an approximation for the resuspension rate, and the net accretion rate (gross accretion less decomposition; Figure 7.16). The generation rates in this balance were estimated from measurements of productivity of the organisms in the water column and from biomass measurements. The striking feature of the mass balance is the large amount of solid material that is cycled, compared to inputs, outputs, or removals. Other studies have produced similar results (Table 7.5).

It may be concluded that in most instances, the effluent TSS from a FWS treatment wetland is determined by

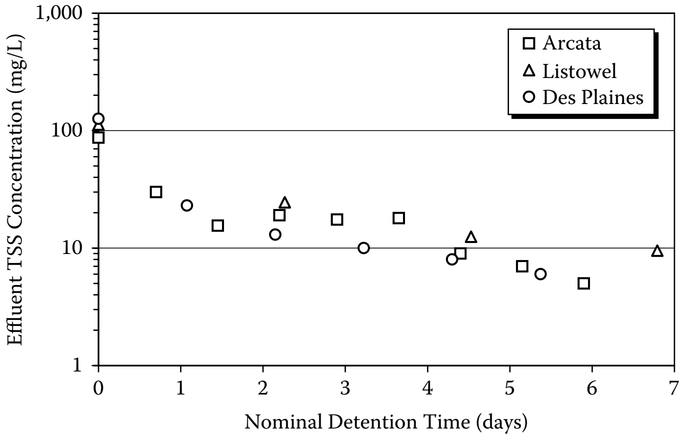


FIGURE 7.15 Gradients in suspended solids along the flow direction in treatment wetlands. (Data for Arcata, California: Gearheart *et al.,* (1989) In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural*. Hammer (Ed.), Lewis Publishers, Chelsea, Michigan, pp. 121–137; data for Listowel, Ontario: Herskowitz, (1986) *Listowel Artificial Marsh Project Report.* Ontario Ministry of the Environment, Water Resources Branch: Toronto, Ontario; data for Des Plaines, Illinois: unpublished data).

internal biological processes, and not by the removal efficiency for incoming TSS. As a corollary, the solids leaving the wetland will very often not be related to the solids entering, but rather to the detrital fragments originating internal to the system.

Seasonal and Stochastic Effects

Because wetland effluent TSS is strongly related to internal ecosystem processes, random physical and biological events have pronounced effects on effluent concentrations. In addition, season and temperature are modifiers of the processes that generate and cycle solids. These effects may be separated by detrending the data, which typically follow a mild annual cycle with superimposed variability. The trend may be determined most accurately if there are data spanning many annual cycles, which may then be "folded" into one multiyear display and averaged.

TSS data time series often display some degree of sinusoidal behavior through the course of a calendar year. Therefore,

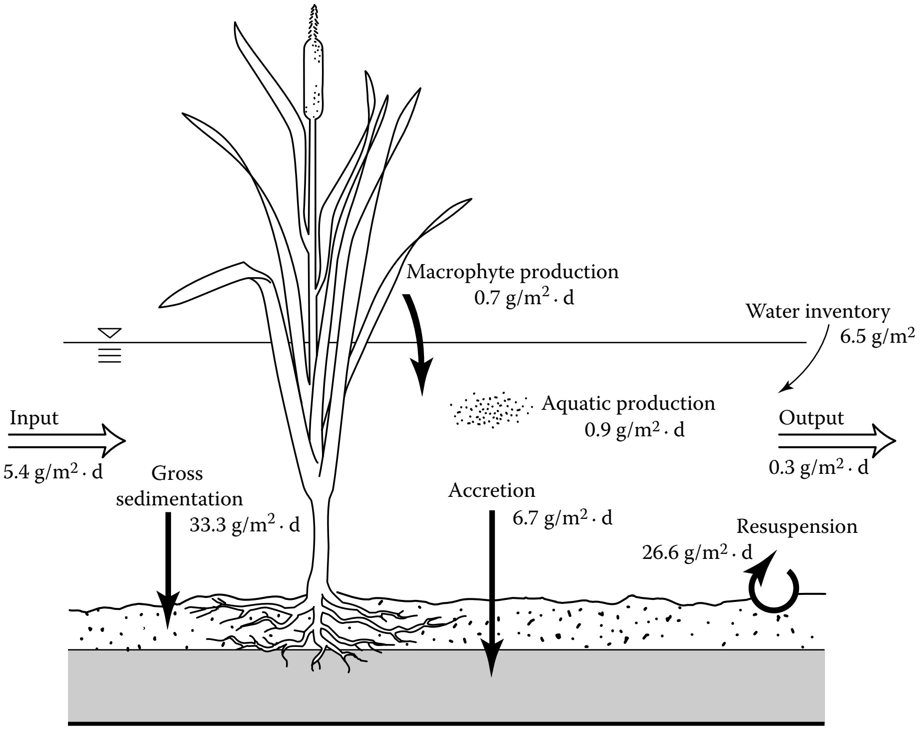


FIGURE 7.16 Components of the sediment mass balance for wetland EW3 at Des Plaines, Illinois. The balance period is the 23-week pumping period in 1991. (Data from WRI (1992) *The Des Plaines River Wetlands Demonstration Project.* Report to U.S. EPA, July 1992. Wetlands Research Inc. (WRI), Chicago, Illinois.) (Adapted from Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

TABLE 7.5
Cycling and Removal of TSS in FWS Wetlands

Site	Inflow ($g/m^2 \cdot d$)	Outflow ($g/m^2 \cdot d$)	Removed ($g/m^2 \cdot d$)	Generation ($g/m^2 \cdot d$)	Cycled ($g/m^2 \cdot d$)
Des Plaines EW3	5.4	0.3	5.1	1.6	26.6
Houghton Lake Pre-discharge	4	1	3	6	53
Olentangy 1	4.7	2.7	2.0	—	95.3
Olentangy 2	4.8	2.7	2.1	—	102.2
Houghton Lake Discharge	13	3	10	60	160

Note: The amounts cycled are far greater than the amounts removed.

Source: Data for Olentangy, Ohio: Harter and Mitsch (2003) *Journal of Environmental Quality* 32(4): 325–334; for Des Plaines, Illinois, and Houghton Lake, Michigan: unpublished data.

detrending may be accomplished by fitting the (folded) time series to

$$C = C_{mean}\left[1 + A\cos\left(\omega(t - t_{max})\right)\right] + E \qquad (7.23)$$

where

A = amplitude fraction
C = concentration, g/m^3 = mg/L
C_{mean} = concentration, g/m^3 = mg/L
E = stochastic departure (error) of an individual measurement, mg/L
t = Julian time, d
t_{max} = Julian time of TSS maximum, d
ω = annual frequency, 2/365, radians/d

The scatter of TSS data is large, and the trend typically accounts for less than 50% of the variability. An example of this model fit to data from the Arcata treatment marshes is given in Figure 7.17, for which $R^2 = 0.26$, implying that only 26% of the variability is accounted by the trend. The amplitude of the annual cycle for Arcata treatment wetlands was 0.32 times the mean. Examples may be found of both weaker and stronger annual trends, as indicated by lesser and greater R^2, with an average for the nine systems in Table 7.6 of $R^2 = 0.20 \pm 0.07$ (mean ± SE).

There is no strong indication of seasonality for the peaks of effluent TSS. These range from winter for Columbia, Missouri; Brighton, Ontario; Imperial, California; and Brawley, California, to autumn for Arcata, California; Cannon Beach, Oregon; and Estevan, Saskatchewan. Listowel, Ontario, peaks in the summer. Outlet peaks correspond only roughly to inlet peak times, with displacements of up to two months. It does not appear that either temperature or season alone is a sufficient predictor of the maximums and minimums of TSS. The temperature coefficient (θ) set forth in Kadlec and Knight (1996) for wetland effluent TSS concentrations was derived from the Listowel, Ontario, data, and appears to be specific for that system. Based on information collected over the last ten years, it is apparent that effluent TSS concentrations vary

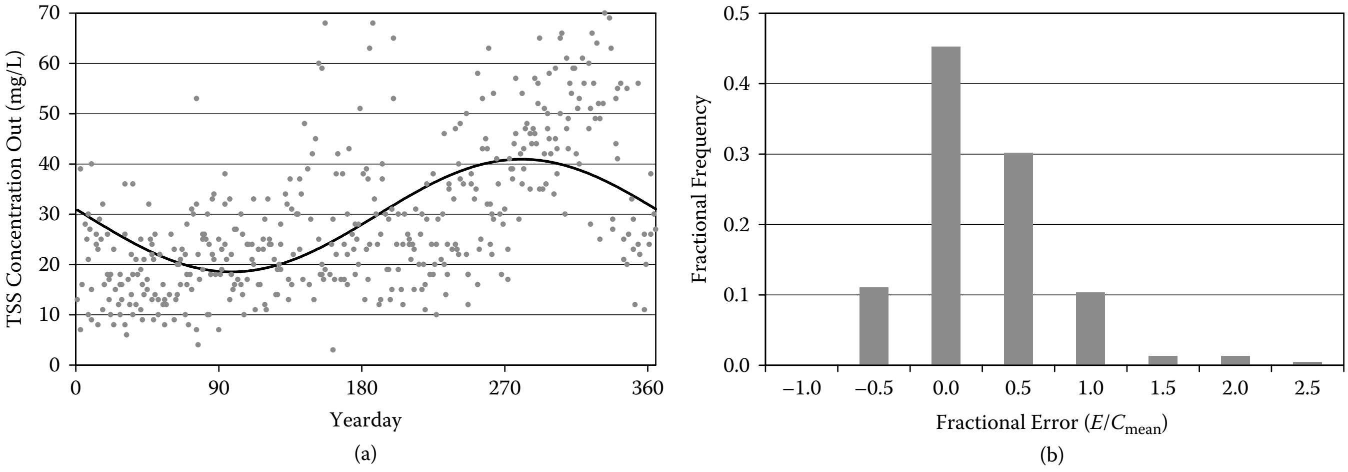


FIGURE 7.17 Suspended solids leaving the Arcata treatment marshes versus day of the year (a). The departures from the sinusoidal trend line extend to 2.5 times the trend values, and are approximately log-normally distributed (b). Thirteen years of weekly data are represented (N = 443). (Data from TWDB database (2000) *Treatment Wetland Database (TWDB).* Website developed for U.S. EPA. http://firehole.humboldt.edu/wetland/twdb.html. Last updated November 2000. Compiled by B. Finney. U.S. EPA: Washington, D.C.)

TABLE 7.6
Annual Trends in Wetland Effluent TSS

Site		Period	Mean (mg/L)	Amplitude Fraction	Max (mg/L)	Min (mg/L)	t_{max} (Julian day)
Arcata, California Treatment	I	Annual	59	0.32	78	40	243
Weekly 13	O		29.7	0.38	41	19	280
Arcata, California Enhancement	I	Annual	27.2	0.24	34	21	284
Weekly 14	O		2.8	0.30	4	2	337
Columbia, Missouri	I	Annual	13.2	0.12	15	12	319
Monthly 3	O		8.1	0.43	12	5	20
Brighton, Ontario	I	Annual	14.3	0.63	23	5	47
Weekly 4	O		7.7	0.32	10	5	27
Imperial, California	I	Annual	35.9	0.16	42	30	116
Weekly 3	O		10.3	0.39	14	6	57
Brawley, California	I	Annual	18.1	0.42	26	10	92
Weekly 3	O		8.1	0.76	14	2	52
Listowel 4, Ontario	I	Annual	111	0.20	133	89	244
Monthly 4	O		7.2	0.64	12	3	176
Cannon Beach, Oregon	I	Dry[a] (summer)	56.0	1.3	71	31	212
Monthly 16	O		6.6	0.16	8	6	218
Estevan, Saskatchewan	I	Summer[b]	21.3	0.84	63	7	330
Weekly 10	O		9.5	0.11	11	9	330

Note: The frequency of sampling is either weekly or monthly as noted. The period record ranges from 3 years (Brawley and Imperial) to 16 years (Cannon Beach). The trend in each time series is presumed to be sinusoidal: $C = C_{mean}(1 + A\cos[\omega(t - t_{max})]) + E$

[a] The means of the full annual cycles are 31.0 and 6.6 mg/L.

[b] The means of the full annual cycles are 41.9 and 10.0 mg/L.

between FWS wetlands. Given this variability in performance response, it can be deduced that performance varies seasonally between FWS wetlands, in ways that are not directly related to temperature. *As a result, it is the current recommendation that no such temperature coefficient be used; essentially,* θ = *1.0 for TSS in FWS wetland systems.*

Because stochastic variability dominates the effluent TSS patterns, that variability requires quantification. For example, in the Arcata treatment marshes, the relative departures from the sinusoidal trend (E/C_{mean}) are approximately log-normally distributed (Figure 7.17). That type of distribution also prevails for other wetland sites, for TSS, and other water quality parameters. This occurs by virtue of the "squeeze" for low data values created by the nearness to the zero level (method detection limit, or MDL) of the parameter (Berthoux and Brown, 2002).

Because wetland effluent TSS distributions are only weakly seasonal, it is possible to ignore these trends, and to lump seasonal effects into the total variability. This is frequently done in the treatment wetland literature (e.g., U.S. EPA, 1999; Wallace and Knight, 2006). The frequency distributions of the inlet and outlet TSS measurements are displayed graphically. Figure 7.18 shows an example of this procedure, derived from the same data as Figure 7.17. Note that the 50th percentile represents the median of the data, not the mean. Further note that these are not paired point graphs, so that reductions cannot be computed at any specified frequency level.

It is useful to examine the multiplier factors associated with the various (higher) percentiles of the effluent distributions, because these may well be involved in permitting or licensing of the treatment wetland. Examples of these outlet multipliers are shown in Table 7.7, for a sampling of wetlands spanning a range of inlet concentrations from 1 to 100 mg/L. It may be seen that in several instances, excursions of outlet concentrations exceed the average inlet concentration, despite long-term average concentration reductions. It is only when the inlet TSS reaches about 25 mg/L that not more than 10% exceedances of the inlet concentration occur.

Input–Output Relations

Suspended solids have been measured at inlets and outlets for a large number of FWS wetlands. It is instructive to examine this large interwetland data set, to ascertain the existence

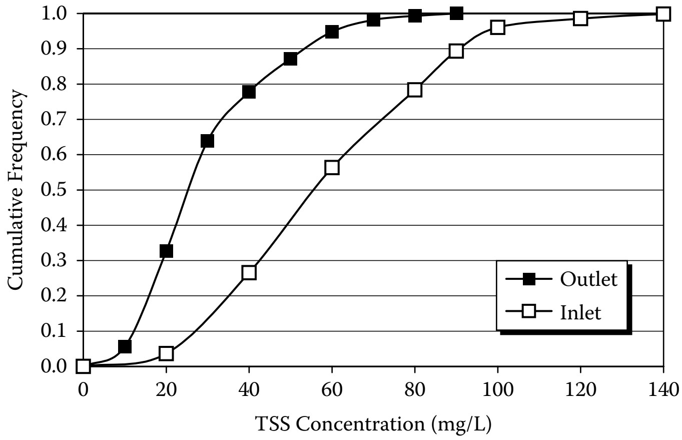


FIGURE 7.18 Probability distributions for inlet and outlet TSS for the Arcata treatment wetlands. The median inlet TSS was 56 mg/L; the median outlet TSS was 25 mg/L. Data were weekly for 13 years. (Data from TWDB database (2000) *Treatment Wetland Database (TWDB).* Website developed for U.S. EPA. http://firehole.humboldt.edu/wetland/twdb.html. Last updated November 2000. Compiled by B. Finney. U.S. EPA: Washington, D.C.)

of trends among systems. A popular method of TSS data representation is the quotation of percentage removal, or removal efficiency. However, the presence of a background TSS level constrains removal efficiency to be below a level dictated by the inlet and background concentrations. As a consequence, percent removal is an inadequate measure for many treatment wetlands. Indeed, some efficiencies are negative, in situations where pretreatment includes removal of TSS prior to the wetland, because influent TSS concentrations are below the wetland background concentrations.

For these reasons, it is preferable to consider graphical exposition of intersystem data, and to derive generalities therefrom. Two choices exist:

1. The input–output concentration graph
2. The outlet concentration–inlet loading graph

Intersystem outlet concentrations apparently increase with the areal loading of TSS to the wetland, with higher outlet concentrations at higher loading rates (Figure 7.19). U.S. EPA (2000a) found a similar pattern for a restricted set of

TABLE 7.7
Trend Multipliers for TSS Distribution of FWS Wetland Effluents

Percentile	Inlet 50 (mg/L)	Outlet 50 (mg/L)	Excursion Frequency 80%	90%	95%	99%
Orlando Easterly, Florida	*1*	*1*	*2.52*	*4.20*	*8.54*	*19.63*
Commerce Township, Michigan	*1*	*9*	*1.60*	*1.96*	*2.44*	*3.62*
Tres Rios, Arizona H1	3	3	2.00	2.33	3.63	5.05
Brighton, Ontario	10	6	2.00	2.50	3.11	6.21
Estevan, Saskatchewan	11	6	2.17	3.33	4.30	9.12
Columbia, Missouri	12	6	1.64	2.26	3.70	5.30
New Hanover, Michigan	13	8	1.33	1.73	1.88	3.86
Brawley, California	18	8	1.69	1.79	1.82	1.83
Listowel, Ontario 3	18	6	1.87	2.72	3.66	4.72
Arcata, California Enhancement	26	3	1.23	1.28	1.29	1.30
Imperial, California	32	10	1.33	1.37	1.39	1.40
Tarrant, Texas WC1	39	5	1.74	2.48	2.76	3.78
Arcata, California Treatment	56	25	1.70	2.12	2.44	2.99
Cannon Beach, Oregon	*58*	*6*	*2.00*	*2.43*	*2.88*	*4.39*
Des Plaines, Illinois EW3	83	7	1.64	1.83	2.09	3.03
Listowel, Ontario 4	*100*	*5*	*1.60*	*3.08*	*3.88*	*5.80*
Mean (ex. Orlando)			**1.70**	**2.21**	**2.75**	**4.16**

Note: The 50th percentile is the median, not the mean. Frequencies are weekly, or monthly (italics). Orlando Easterly data is strongly left-censored, with an MDL of 1.0 mg/L. Trend multiplier is $(1+\Psi)$; see Equation 6.61.

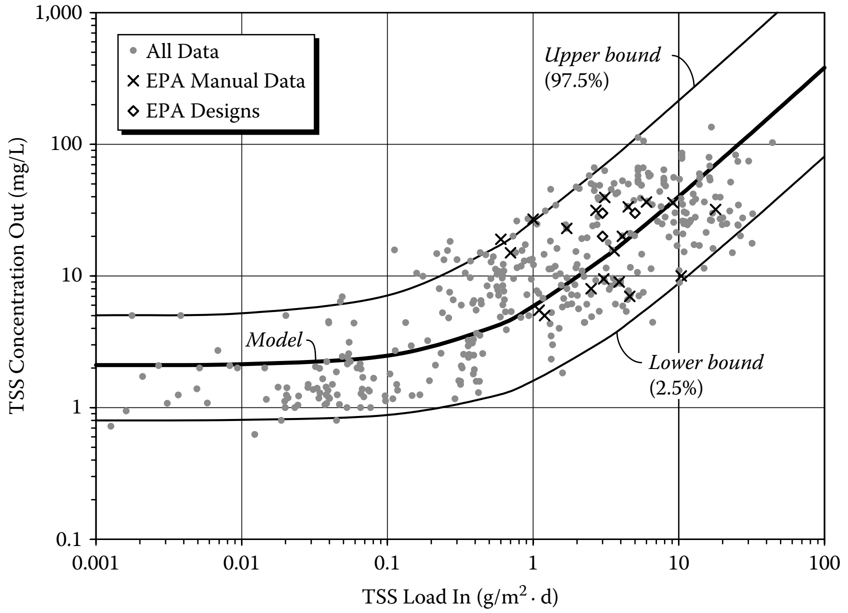


FIGURE 7.19 Load response for TSS in FWS wetlands. Data represent one point for one wetland for one year. N = 388, for 136 wetlands. Diamonds represent EPA design recommendations (U.S. EPA, 2000a). The central line is a linear regression, with $R^2 = 0.65$ for the logarithmic basis shown. The upper line represents 97.5% bound of the data; the lower line represents the 2.5% bound. Fitting parameters for TSS = (A + (B × Loading)) are:

	A (mg/L)	B (mg/L)/(g/m²·d)
Upper bound	5	21
Central tendency	2.1	3.8
Lower bound	0.8	0.8

data, which are also shown on Figure 7.19. At any given loading rate, the data cloud spans about a factor of 10 in outlet concentrations. The central tendency and upper and lower bounds are shown, together with the corresponding regression equations. However, this view of system performance is very misleading.

When data from a given site are examined, a different picture emerges. Figure 7.20 shows results from six different side-by-side tests at four locations, each of a year or more duration. Different TSS loadings were achieved by varying the hydraulic loading. Depth, source water, and meteorology and other site factors were invariant within each group of data. In each group, the spread of the inlet TSS loadings was a factor of 5–10. An interesting and important observation is that there is essentially no increase in outlet TSS with TSS loading within each group. Therefore, *TSS loading is an inappropriate correlating parameter for prediction of outlet TSS.*

This means the *k-C* model* (*as described in Chapter 6*) *is dominated by C**. For the *k-C** model, we expect to see an "S" curve on the loading graph, with C^* as one asymptote and C_i as the other asymptote. In contrast, the FWS wetlands analyzed in Figure 7.19 never approached C_i as the hydraulic retention time (τ) was decreased; the wetlands continued to return an outlet TSS concentration that is a function of internal TSS generation ($G + R$), which is represented by C^*.

Other factors most responsible for the large differences in outlet TSS concentrations at the same inlet loading and the likely candidates are inlet TSS concentration and inlet nutrient status. These two factors often go hand-in-hand, and there is not yet a study that has identified the relative importance. High inlet TSS concentrations could be partially short-circuited to the wetland outlet, or high nutrients could cause more internal generation of TSS.

At this point in the history of treatment wetland technology, we are only left with the possibility of input–output regression relationships to predict output TSS concentrations. A large intersystem data set for annual values is shown in Figure 7.21, together with regression lines for the central tendency, and upper and lower bounds set to confine the middle 95% of the data. Regression of the annual information produces the following correlation:

$$C^* = C_o = 1.5 + 0.22\, C_i \tag{7.24}$$

where

$R^2 = 0.65$ for logarithmic data, N = 443
$0.2 < C_i < 1{,}910$ mg/L
$0.6 < C_o < 135$ mg/L

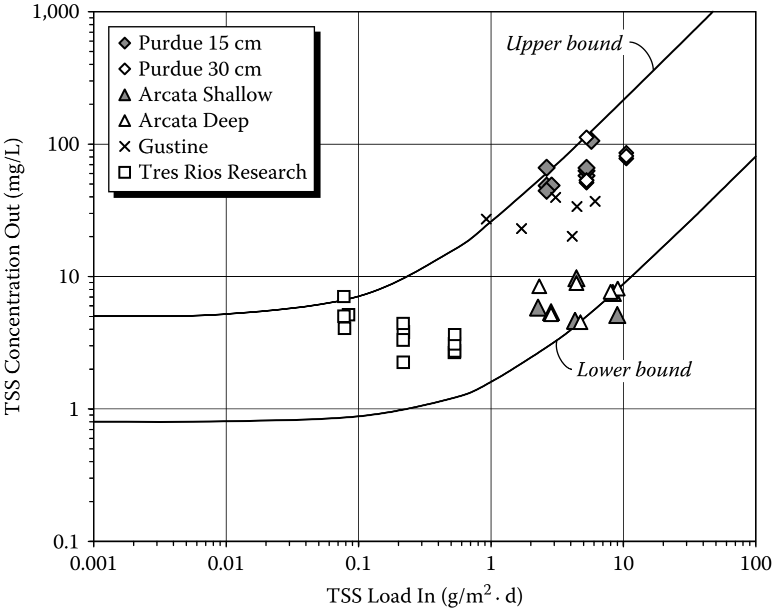


FIGURE 7.20 Intrasystem behavior of FWS wetlands in response to loading changes. Each point represents one year of data, and all cell clusters were run side by side.

Site	No. Cells	Depth (cm)	HLR (cm/d)	TSS Inlet (mg/L)	NH_3-N Inlet (mg/L)
Purdue 15 cm	8	15	2–4	135–145	400
Purdue 30 cm	7	30	4–8	135–145	400
Gustine	6	45	1–4	75–150	14–20
Arcata Shallow	6	30–40	6–25	36	9–14
Arcata Deep	6	49–61	6–25	36	9–14
Tres Rios Research	12	41–62	2–15	4	2

Open Water Areas

Curiously, the subject of inclusion of open water areas in FWS treatment wetland systems has been bifurcated into

1. Deep zones inside the wetland
2. Ponds preceding wetlands

An inlet deep zone inside the wetland is essentially a pond located inside the wetland boundary. Ponds function to settle incoming TSS, but are conducive to the production of TSS via algal cycling, as discussed in Chapter 3. Internal and outlet open water areas are settling zones, but are subject to wind resuspension and algal growth.

U.S. EPA (2000a) suggests that open zones be incorporated into treatment wetlands as a means of enhancing TSS removal, along with other purposes. The reason given is that these "…can provide conditioning and transformation processes which may improve overall removal of TSS…." "Open zones" may contain submerged vegetation, or be devoid of plants. This differentiation is very important, because open zones with submerged aquatic vegetation (SAV) will provide TSS reduction benefits, whereas unvegetated open zones will not, and may in fact increase TSS.

Pond–Wetland Combinations

Because incoming TSS is rapidly settled and filtered in the wetland environment, it is possible and desirable to provide a first element of the treatment wetland complex that traps the fastest settling fraction of the suspended material. A pond provides for that presettling and is more easily cleaned than an emergent or submergent macrophyte bed. It is further desirable to collect solids and their partitioned metals and chemicals in a location that is not foraged by sediment-feeding vertebrates. This presettling pond may require infrequent dredging to remove the accumulated deposits.

Data from the Tarrant County, Texas, site (APAI, 1995) illustrates the mean performance of three parallel marsh wetland cell trains of three cells each, following two parallel unvegetated settling ponds. The settling ponds occupied 15% of the area, but accounted for 94%–97% of the solids removal (Figure 7.22). The first wetland cell completes the solids removal; the remaining two cells do not reduce TSS any further. The last wetland cells were, however, needed for phosphorus removal. The performance of five pond–wetland systems is summarized in Table 7.8. The large majority of

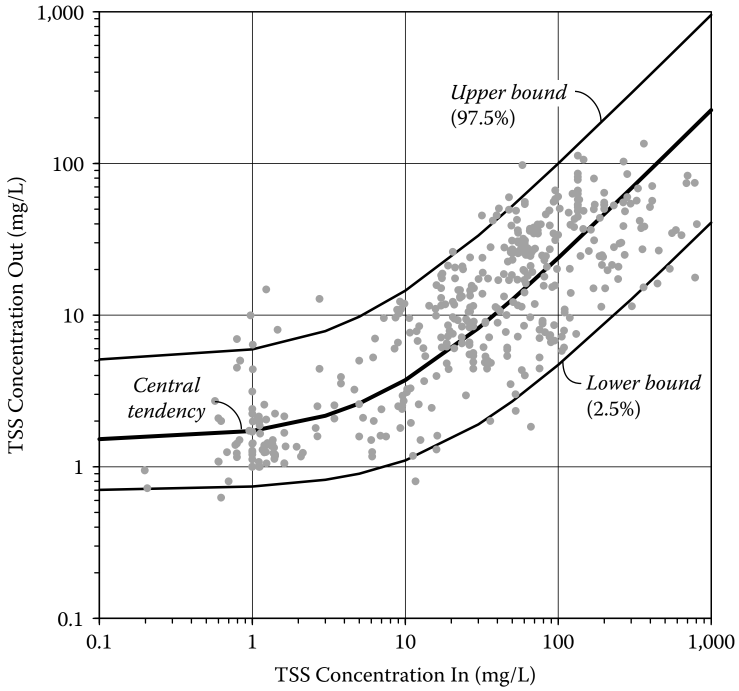


FIGURE 7.21 Input–output plot for TSS in FWS wetlands. Data represent one point for one wetland for one year, N = 443, for 142 wetlands. The central line is a linear regression, with $R^2 = 0.65$ for the logarithmic basis shown. The upper line represents 97.5% bound of the data; the lower line represents the 2.5% bound. Fitting parameters for $C_o = (A + (B \times C_i))$ are:

	A (mg/L)	B (Dimensionless)
Upper bound	5	0.95
Central tendency	1.5	0.22
Lower bound	0.7	0.04

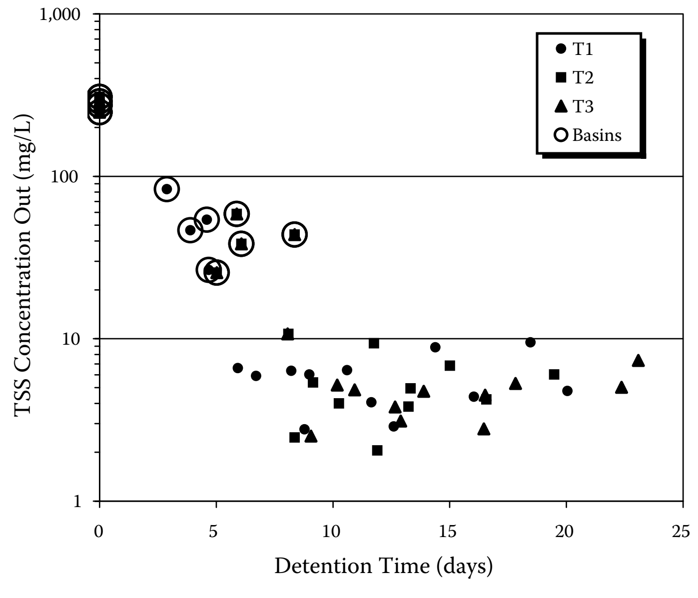


FIGURE 7.22 Profiles of TSS along the flow direction in the Tarrant treatment trains over a four-year period. Note that the wetlands exhibit a plateau, or background TSS, below 10 mg/L. (From APAI (1995) *The use of constructed wetlands for protection of water quality in water supply reservoirs.* Final report by APAI (Alan Plummer and Associates, Inc.) to the American Water Works Association Research Foundation and the Tarrant County Water Control and Improvement District No. 1, AWWA: Denver, Colorado.)

TABLE 7.8
TSS Removal in Systems with a Presettling Basin Followed by a Wetland

Site	Sed Basin (% Area)	TSS In (mg/L)	Sedimentation Basin TSS Out (mg/L)	Sedimentation Basin Load Removed (g/m²·yr)	Wetland TSS Out (mg/L)	Wetland Load Removed (g/m²·yr)
Tarrant, Texas 1	12	276	46	20,570	6	1,312
Tarrant, Texas 2	15	276	37	21,993	11	1,181
Tarrant, Texas 3	15	276	28	22,871	6	742
Brawley, California	25	216	35	21,585	12	858
Imperial, California	≈30	200	18	10,055	7	1,418

Note: All systems were run for more than three years, and had four to seven days' detention in the sedimentation basins.

TSS in these systems was retained in a minority fraction of the total footprint.

The placement of a pond as the final element in a wetland treatment system is generally not desirable from the standpoint of TSS reduction. The planktonic production in such a pond is typically quite high, leading to the reintroduction of high-chlorophyll microdetritus, much of which remains in suspension. An example of this phenomenon was the Lakeland, Florida, system. Entering TSS was reduced in the first marsh cells but was regenerated in later, open water cells because of planktonic activity (Bays *et al.*, 1993).

Deep and Open Water: Unvegetated Zones

A treatment wetland may include an inlet deep zone, within the footprint of the wetland. Such a feature is in effect a pretreatment pond element, as discussed in the previous section. In contrast, the wetland may also contain internal deep zones, ranging from narrow ditches to large expanses of open water. Both the number and size of such zones have been variable across a number of projects. Here, we examine the efficacy of such zones in TSS reduction.

Moore and Niswander (1997) operated a set of six treatment wetlands receiving diluted dairy wastewater for two years. Two wetlands had a central internal deep zone comprising 45% of the area, whereas the other four had no internal deep zone. Data for the second year, past the start-up period, were analyzed for differences in performance. The wetland areas were all identical, and hence the hydraulic loading to all six wetlands was uniform at 3.95 cm/d. For the two cells with deep zones, the average TSS influent of 653 mg/L was reduced to 202 mg/L For the four cells without deep zones, the average TSS influent of 653 mg/L was reduced to 195 mg/L. The authors concluded that deep center sections did not show any significant impact on treatment efficiency.

Knight *et al.* (1994) operated a set of six treatment wetlands receiving treated paper pulp mill effluent for two years. Two internal deep zones comprising 25%, 35%, and 45% of the area were included in three of the six. There was variability in flow (different HLRs) and aspect ratios (2.5, 5, and 10). If the pairs of cells with the same aspect are compared on the basis of deep zones, the concentrations produced were lower, and the load removed was higher, in wetlands with deep zones in two of the three pairs.

Eidson *et al.* (2005) studied side-by-side wetlands at Augusta, Georgia. Six 12-ha cells were studied, two of a marsh–pond–marsh configuration (60% marsh), and four of a marsh–ditch configuration (82% marsh). Incoming water was very low in TSS, averaging 3.3 mg/L over a one-year period. The marsh–pond–marsh cells averaged 12.2 mg/L at their outlets, whereas the marsh–ditch cells averaged 4.1 mg/L. The effect of the large open water areas in the marsh–pond–march cells was to raise the TSS concentration, presumably due to generation and resuspension. The marsh–ditch systems created a negligible increase.

At the Tres Rios, Arizona, project, Kadlec (2007) studied the effect of deep zones in a triplicated side-by-side study. The research site contained a set of twelve 0.12-ha (24 m wide × 50 m long) wetlands, built in a triplicate design with 2, 3, 4, and 5 deep zones, including one at the inlet and one at the outlet. No significant difference could be found in outlet TSS concentrations ($\alpha = 0.05$).

Further insights can be gained from systems that lost their vegetation over the course of time. A FWS treatment wetland in Commerce Township, Michigan, was "eaten out" by muskrats and waterfowl, leaving virtually no emergent or submergent vegetation. Before the loss, the effluent TSS was 5 mg/L; after, it was 13 mg/L.

The Tres Rios Hayfield wetlands are a second example (Kadlec, 2006). During spring and summer of the third year after start-up, the vegetation essentially all died, for reasons that have not been resolved, and regrowth did not occur. Incoming TSS was low (3 mg/L), and remained relatively low during the vegetative period (2 mg/L) (Figure 7.23). After loss of vegetation, effluent TSS climbed to 27 mg/L. This wetland had unvegetated deep zones. In both cases it is apparent that the effluent solids could not have derived from incoming TSS, but rather were the result of internal generation and resuspension.

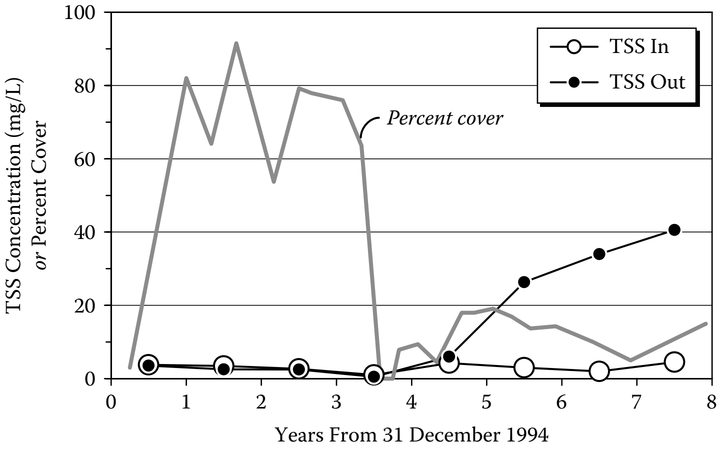


FIGURE 7.23 The transition of Tres Rios wetland H1 from a vegetated to an unvegetated state. Outlet TSS averaged 2 mg/L for the first four years, then climbed to 40 mg/L in year eight.

Submerged Aquatic Vegetation (SAV)

It is well known that SAV reduces resuspension in lake environments (James and Barko, 2000; James *et al*., 2001; 2002; Horppila and Nurminen, 2003). In shallower wetland environments, SAV would presumably serve that same function, and would provide the additional benefits ascribable to the submerged portions of emergent vegetation. Consequently, TSS will be generated in SAV systems.

Very few studies of TSS behavior in SAV beds have been reported. DB Environmental (DBE, 1999) measured 2 mg/L in, and 3 mg/L out, of SAV mesocosms treating agricultural runoff. However, the annual accretion rate of new sediments was 1.0 cm/yr, again indicating that internal generation was a dominant mechanism. Toet (2003) measured turbidity in a set of nine side-by-side wetlands receiving highly treated municipal effluent. The front halves were vegetated with *Typha* (4) and *Phragmites* (4). The back halves were vegetated with SAV (*Elodea*, *Ceratophyllum*, and *Potamogeton*), and there was an open water control. The front sections increased turbidity from 3 to 6 NTU, and the back sections provided a slight further increase to 7 NTU.

Based upon this limited wetland information, it appears that SAV beds have about the same background TSS as emergent wetlands; but upon lake information, it is expected that SAV will help prevent resuspension. Overall, the current expectation is that SAV beds will behave approximately like emergent systems for TSS processing.

7.4 PARTICULATE PROCESSES IN HSSF WETLANDS

Although HSSF wetlands are configured very differently than FWS wetlands, the same physical processes apply to different degrees and to different magnitudes. Processes that affect the removal and generation of particulate matter in HSSF wetlands are discussed in this section. Like FWS wetlands, HSSF wetlands are very effective in trapping and retaining TSS associated with the inlet flows. Unlike FWS wetlands, this accumulated TSS material reduces the hydraulic conductivity of the wetland, often to a significant degree. Bed clogging that occurs in HSSF wetlands as a result of TSS accumulation has often led to hydraulic failure and associated flooding of the wetland bed, which remains a significant operation and maintenance challenge to this day.

Particulate Settling

Like FWS wetlands, HSSF wetlands are very effective at removing TSS associated with the inlet flow. One of the primary mechanisms is gravitationally driven particulate settling. This has already been discussed in detail for FWS wetlands (Equations 7.1–7.7). Because the bed porosity in HSSF wetlands is low (ε = 0.30–0.40) relative to FWS wetlands, it is useful to consider gravitational settling in terms of the actual flow velocity (v) rather than the superficial flow velocity (u). Thus, Equation 7.6 can be rewritten as

$$t_{\text{travel}} = \frac{L}{v} \tag{7.25}$$

where

L = wetland length, m
t_{travel} = time to traverse wetland, s
v = actual flow velocity, m/s ($v = u/\varepsilon$)
u = superficial flow velocity, m/s
ε = bed porosity, dimensionless

Theoretically, all particles of a size corresponding to a given fall velocity will be removed by settling if the travel time exceeds the settling time. In FWS wetlands, the *fall distance* is approximated as the overall water depth within the wetland. In HSSF systems, the wetland is filled with a granular bed. The porosity of this bed increases the flow velocity ($v > u$), but decreases the fall distance, because the particle only has to fall the distance of the average pore space before hitting an intercepting surface, not the entire depth of the wetland bed. In most instances, the pore size within a HSSF wetland bed can be approximated by the d_{10} of the bed media (90% of the particles within the bed are larger than the d_{10}). Thus, Equation 7.7 can be rewritten as

when

$$\frac{L}{v} > \frac{d_{10}}{w}$$

$$N_{\text{fall}} = \frac{Lw}{vd_{10}} > 1 \tag{7.26}$$

where
L = wetland length, m
v = actual flow velocity, m/s
d_{10} = particle size representing the smallest 10% of the bed media
w = terminal solids settling velocity, m/s
N_{fall} = particle falling number, dimensionless

As a practical matter, generally the falling rate (w) is much greater than the actual flow velocity (v), ($w >> v$). As a result, virtually all the particles associated with the influent wastewater are settled out, generally within the first 5% of the wetland bed (Puigagut *et al.*, 2006).

Filtration and Interception

As discussed for FWS wetlands, the principal mechanisms of granular bed filtration are well known and documented in handbooks (see, e.g., Metcalf and Eddy Inc., 1991; Crites and Tchobanoglous, 1998). These include:

1. Inertial deposition, or impaction—particles moving fast enough that they impact bed particles rather than being swept past by the flowing water.
2. Diffusional deposition—random processes at either microscale (Brownian motion) or macroscale (bioturbation) which move a particle to an immersed surface.
3. Flow line interception—particles moving with the water and avoiding head-on collisions, but passing close enough to graze the stem and its biofilm, and sticking.

Media size in HSSF wetlands around the world ranges from soils (d_{10} < 0.1 mm) up to coarse gravels (d_{10} > 4 mm). This size range in bed media spans the dominant scale factors of Mechanisms 1–3 listed above. For fine-grained bed media, Mechanisms 1 and 2 will predominate. For gravel media, Mechanism 3 will be the most important.

As a practical matter, these mechanisms all combine to preferentially remove incoming TSS in the inlet region of the HSSF bed. For fine-grained media, Mechanisms 1 and 2 remove particles almost immediately. In coarser bed (gravel) systems, Mechanism 3 will predominate, and will work in conjunction with the particulate settling mechanisms just described.

Resuspension

In contrast to FWS wetlands, resuspension mechanisms are strongly minimized in HSSF wetlands due to the physical configuration of the HSSF reactor. Flow velocities within the HSSF bed are low, and generally do not generate shear stresses sufficient to scour particulate matter. As flow in HSSF wetlands occurs below the top of the bed, resuspension mechanisms such as wind mixing and turbulence are not factors. Similarly, bioturbation (from burrowing rodents) and gas lift, although theoretically possible, occur at such small localized scales, that their effect on the overall wetland is nil. As a result of these factors, resuspension is generally not a significant phenomenon in HSSF wetlands.

Chemical Precipitation

Reaction chemistry as noted previously for FWS wetlands can also occur in HSSF wetlands. One use of HSSF wetlands has been as sulfate-reducing systems to induce the precipitation of copper, nickel, and other metals (Eger, 1992). Many metals form highly insoluble sulfide precipitates (Palmer *et al.*, 1988), as discussed in Chapter 11. A peat-bed HSSF wetland has been used since 1986 to remove copper and nickel from mine drainage at the LTV Dunka Mine near Hoyt Lakes, Minnesota (Eger and Lapakko, 1989; Frostman, 1993).

Other than HSSF wetlands treating mine wastes (Younger *et al.*, 2002), accumulation of chemical precipitates generally does not occur at a rate significant enough to impact the hydraulic conductivity of the HSSF wetland bed.

Production of Biological Solids

Although HSSF wetlands are effective in removing influent suspended solids through settling, interception, and filtration, and may generate small amounts of solids through chemical precipitation, the majority of the particulate matter present in a HSSF bed treating primary or secondary domestic wastewater consists of biological solids that are generated internally within the system. These consist of

1. Plant detrital material (including associated microbial and fungal networks)
2. Microbial films present on bed media particles

Plant Contributions

Cumulative experience with HSSF wetlands indicates that deeper gravel beds (>40 cm) will contain an upper zone that contains essentially all the plant roots and a lower zone without roots. The presence of root blockage is an important factor: the root zone impedes flow more than the relatively clean media below it. Several tracer studies have documented this phenomenon (Fisher, 1990; Pilgrim *et al*., 1992; Tanner and Sukias, 1995; Marsteiner *et al*., 1996; Tanner *et al*., 1998a; Drizo *et al*., 2000; García, 2003).

Apart from the living root and rhizome material, the upper layer of an HSSF wetland may contain significant amounts of organic matter associated with the plants. For example, Tanner and Sukias (1995) found that planted wetlands developed at least twice as much organic matter in the top 10 cm, compared to unplanted replicates, over a 22-month period. It was not determined whether this material was generated by above- or belowground plant activity.

Microbial Contributions

The solids in HSSF wetlands originate from particulates (filtration) and from living and dead microbial biomass (biosolids = sludge). Microbial biomass forms in response to both particulate and soluble organic loading rates. These biofilms further entrap both organic and inorganic solids (Winter and Goetz, 2003), forming a composite material. In soil absorption systems, this material is contained in a layer commonly termed a *biomat* (Crites and Tchobanoglous, 1998; Beal *et al*., 2004). Others have designated it as *sludge* (Cooper *et al*., 2006a) or *biosolids* (Ragusa *et al*., 2004). Internal solids accumulation can also be affected by chemical phenomena such as sulfide precipitation (Liebowitz *et al*., 2000), and varies in different applications depending on the nature of the waste being treated. Acknowledging that wetlands internal solids are often mostly organic, and are spatially distributed in at least two dimensions, we opt for calling these internal bed materials *biosolids*.

Biosolids formation is greatest at the inlet end of the wetland where the organic loading is highest (Ragusa *et al*., 2004). The loss of pore volume due to biomat formation reduces the hydraulic conductivity in this inlet zone (Zhao *et al*., 2004) (see Chapter 2). Organic matter is removed as wastewater flows through the wetland, resulting in declining biosolids growth. At the outlet, where only small quantities of soluble organic matter are available to the microbes and fungi, biosolids formation is minimal. The nonuniform distribution of internal biosolids along the length of the bed results in a nonuniform distribution of hydraulic conductivity throughout the bed, as discussed in Chapter 2.

Accretion and Bed Clogging

The combined effects of particulate settling, filtration, and interception result in highly efficient trapping of TSS within the inlet region of the HSSF bed. Wetland plants root preferentially within the upper regions of the HSSF bed, obstructing flow in this region. The loading of organic matter, for systems treating domestic wastewater, in both soluble and particulate forms, results in the preferential development of microbial biomats in the inlet region of the HSSF wetland bed. The net result of these mechanisms is a highly nonuniform distribution of solids, plant roots, microbial activity, and associated reductions in hydraulic conductivity, as discussed in Chapter 2. Eventually, this inlet zone may become clogged, and the bed will develop overland flow in this region.

Clogging can occur just from deposited particulate (mineral + organic) material. In a laboratory experiment, Sun (1998) was able to demonstrate that when enough sawdust was added to a flume containing pea rock (effectively reducing the porosity from 39.5% to 33.4%), the resulting head loss was controlled by the particulate matter, not the bed media (Sun *et al*., 1998). Porosity reduction due to particle trapping provides reasonable estimates of the time to clogging (Blazejewski and Murat–Blazejewska, 1997).

Most organic matter is removed in the inlet zone of the HSSF wetland bed. This is the zone of heaviest biosolids accumulation, where the greatest reductions in hydraulic conductivity occur. This zone can be termed the *biosolids clogging distance* and is analogous to the clogging mat that develops in soil infiltration systems treating septic tank effluent (U.S. EPA, 2002c). A schematic of the clogging phenomenon is shown in Figure 7.24.

In fine-grained materials, there is greater bed particle surface area available per unit length of flow path. As a result, more microbial biofilm can form in response to the organic loading. Because the pore size is smaller, the biosolids are more effective in entrapping organic and inorganic solids (as discussed under the Filtration and Interception section above). If the resulting accumulation completely fills the pore spaces, the hydraulic conductivity is controlled (reduced) by the characteristics of the biosolids and not by the characteristics of the media. In this case, the wastewater will likely surface. Consequently, fine-grained media such as HSSF soil filters are unlikely to avoid clogging and the associated flooding, and overland flow.

With coarse bed materials, there is less surface area available for biofilm formation per unit length of flow path. Due to the larger pore spaces, the biosolids cannot completely fill the pore volume, and effective flow paths through the media still exist. The net effect lengthens the biosolids penetration distance but decreases the potential for plugging (Zhao *et al*., 2004). This concept is illustrated in Figure 7.24.

Progressive accumulations of biosolids can lead to a progressive clogging failure of the wetland bed, and the HSSF wetland will end up functioning as an overland flow treatment system, as illustrated in Figure 7.25. This mode of hydraulic failure has occurred in many HSSF wetlands (see Figure 2.28).

7.5 TSS REMOVAL IN HSSF WETLANDS

The fate and transport of TSS in HSSF wetlands are under the same physical principles as in FWS wetlands. Consequently, Equations 7.15–7.21 can be applied to HSSF wetlands, with the exception that the actual flow velocity (v) is used instead of the superficial flow velocity (u). *Note,* $v = u/\varepsilon$*, where* ε

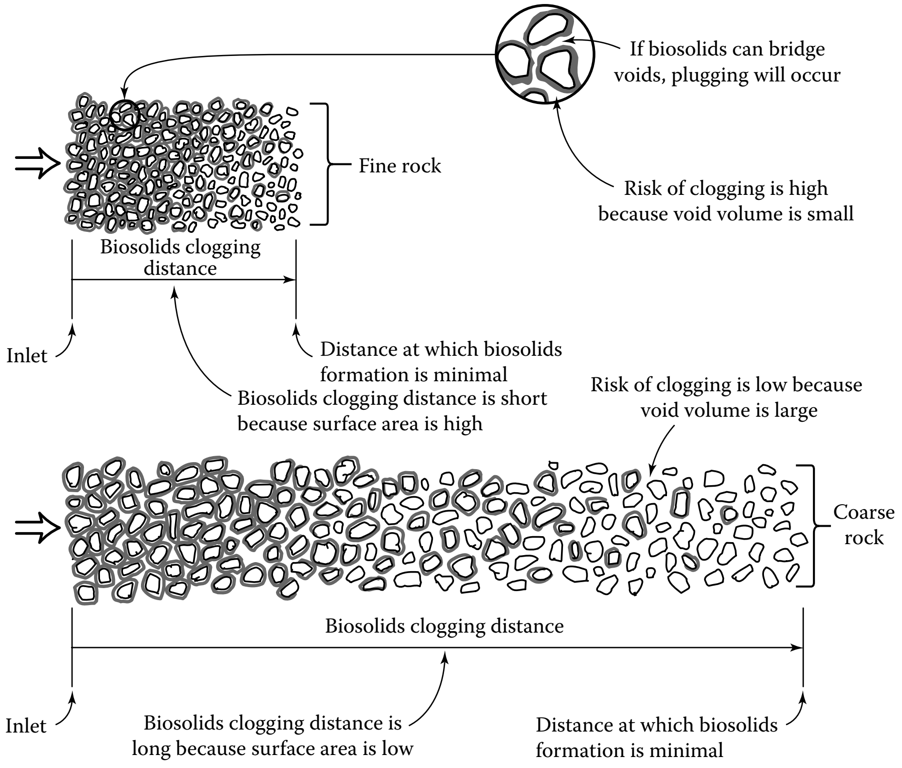


FIGURE 7.24 Biosolids clogging distance as a function of media size. (From Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia. Reprinted with permission.)

represents the HSSF bed porosity (ε *is typically between 0.3 and 0.4*).

Application of these equations is of limited utility, as in many instances the TSS entering the wetland is removed very rapidly, and the effluent TSS leaving the wetland is determined by internal biological processes, but not by the removal efficiency for incoming TSS, as indicated in Figure 7.26. However, if HSSF wetland detention times are small, settling may not be complete, and the w-C^* model (Equations 7.20 and 7.21) provides a reasonable description (Figure 7.27). As a result, the solids leaving a HSSF wetland are typically not related to the solids entering the system, but are produced by the decomposition and resuspension of biomat particulates within the HSSF bed.

Seasonal and Stochastic Effects

Because HSSF wetland effluent TSS is a function of internal ecosystem processes, random physical and biological events result in short-term effects on effluent concentrations. Additionally, season and temperature are modifiers of the processes that generate and cycle solids. These effects typically follow a mild annual cycle with superimposed variability.

The trend may be determined most accurately if there are data spanning many annual cycles, which may then be "folded" into one multiyear display and averaged. TSS data time series often display some degree of sinusoidal behavior through the course of a calendar year, as described by Equation 7.22. An example of this is shown in Figure 7.28.

Examples of sinusoidal fitting of seasonal behavior in HSSF wetlands are summarized in Table 7.9 for three tertiary and four secondary treatment wetlands in England. The scatter of TSS data is large, and the seasonal trend typically accounts for a small percentage of the variability. For the tertiary HSSF wetlands listed in Table 7.9, effluent R^2-values range from 0.02 to 0.12, implying that 2%–12% of the effluent TSS variability could be attributed to seasonal effects. For the secondary systems listed in Table 7.9, effluent R^2-values range from 0.02 to 0.20. In general, these R^2-values are lower than for FWS wetlands.

HSSF wetlands typically display a peak TSS concentration in spring or summer (t_{max} between 112 and 198 days for the Northern Hemisphere), as shown in Table 7.9, but these seasonal peaks account for only a small fraction of the effluent variability in TSS.

Because TSS concentrations in HSSF wetlands are only weakly seasonal, it is possible to ignore seasonal effects, and combine this into the overall variability of the system. This has been done in the treatment wetland literature (Wallace and Knight, 2006) for combined wetland data sets. Figure 7.29 shows an example of this approach.

Based on the probability distribution of effluent TSS concentrations, it is possible to determine a multiplier (C_o/C_{median}) associated with a given percentile of the effluent distribution.

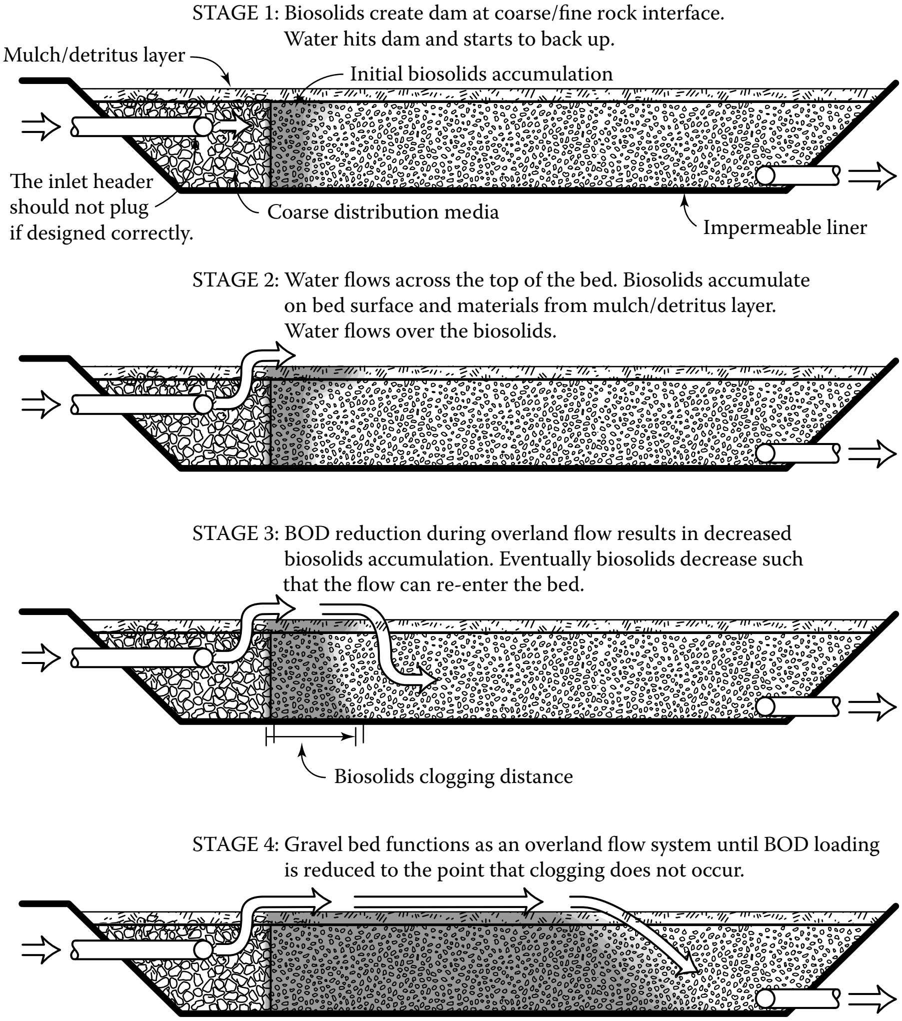


FIGURE 7.25 Stages of bed clogging in HSSF wetlands. (Adapted from Wallace and Knight (2006) *Small-scale constructed wetland treatment systems: Feasibility, design criteria, and O&M requirements*. Final Report, Project 01-CTS-5, Water Environment Research Foundation (WERF): Alexandria, Virginia; and D.J. Cooper *et al.* (2006b) *Factors affecting the longevity of subsurface horizontal flow systems operating as tertiary treatment for sewage effluent*: *Part II*. Kröpfelová (Ed.), Paper presented at the 6th International Workshop on Nutrient Cycling and Retention in Natural and Constructed Wetlands, 31 May–4 June 2006; Trebon, Czech Republic.)

These multipliers are often useful because they may well be involved in the permitting or licensing of a treatment wetland. Examples of these effluent multipliers are shown in Table 7.10 for seven HSSF wetlands (three tertiary and four secondary) in England. Lightly loaded (tertiary) HSSF wetlands often return a median effluent concentration close to detection limits. These systems display greater effluent variability than more heavily loaded systems with higher effluent TSS values. This greater variability can be attributed to stochastic effects, especially sampling error, that result in isolated high instances of effluent TSS. These instances still occur in secondary HSSF treatment wetlands, but the impact is not as great because these wetlands return, on average, a higher effluent TSS to begin with.

Input–Output Relations

Suspended solids have been measured at inlets and outlets for a large number of HSSF wetlands. As the TSS leaving the wetland is a function of internal biological processes, there is a nonzero background concentration (C^*) for effluent TSS. HSSF data sets can be explored graphically; the most commonly used relationship is the outlet concentration (C_o) versus inlet loading graph (Figure 7.30).

As has been previously discussed, outlet concentrations (C_o) are generally not related to inlet concentration (C_i) (Figure 7.26). Also, outlet TSS concentrations have only minor seasonal variations (Figure 7.28). Therefore, there is little apparent relationship between TSS loading and effluent TSS concentrations. The data presented in Figure 7.30 indicate that HSSF wetlands return an average effluent concentration of 22.5 mg/L over a wide range of influent loadings, with a 90th percentile limit of 42 mg/L. This is broadly consistent with other statistical analyses of TSS removal in HSSF wetlands (Wallace and Knight, 2006). Performance criteria postulated by the U.S. Environmental Protection Agency (U.S. EPA, 2000a) also fall within the 50th and 90th percentile bands shown in Figure 7.30.

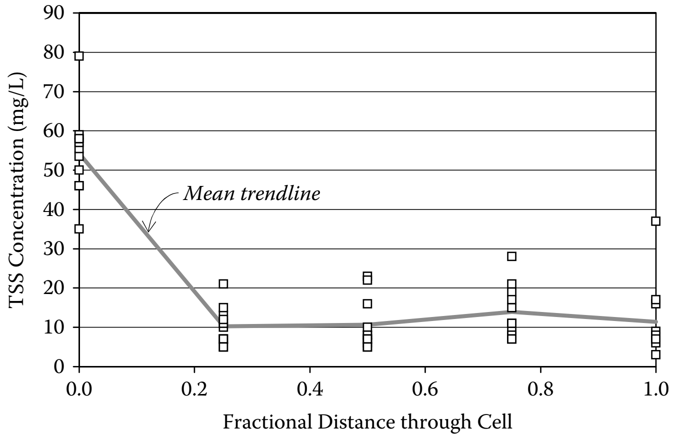


FIGURE 7.26 Internal profile of TSS for the NERCC wetland near Duluth, Minnesota. Data represents eleven transects taken down the length of the HSSF wetland bed between May and October 1998. The hydraulic loading was small, 1.1–1.3 cm/d, and hence nominal detention times were about 14 days. Essentially all the inlet TSS is removed prior to the first sampling port (25% down the length of the bed). (From unpublished data.)

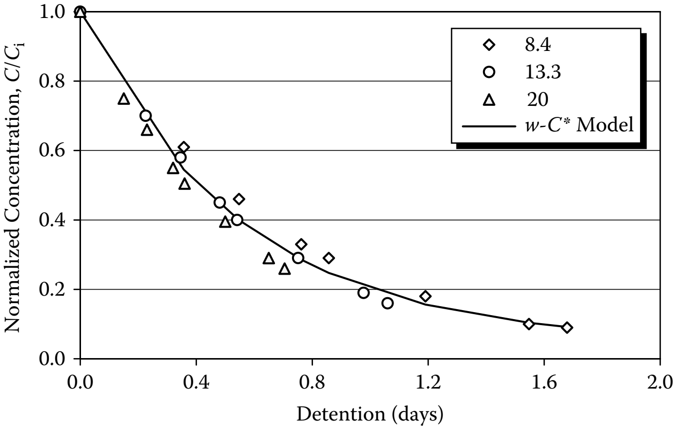


FIGURE 7.27 Turbidity reduction for a HSSF wetland at Richmond, New South Wales, Australia, at different superficial velocities (8.4–20 m/d). The settling rate is $w = 0.29$ m/d, and the background is $C^*/C_i = 0.046$ ($R^2 = 0.987$). (Data from Sapkota and Bavor (1994) *Water Science and Technology* 29(4): 55–66.)

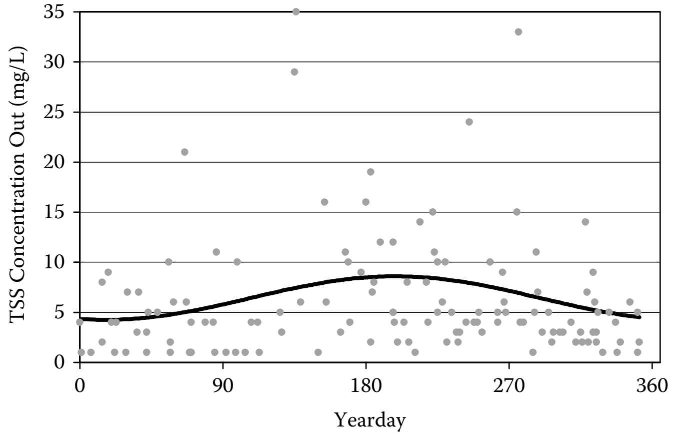


FIGURE 7.28 TSS effluent concentrations for a HSSF treatment wetland in Staffordshire, England. Four years of data are represented; sampling frequency was every two weeks. (Data from CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

TABLE 7.9
Annual Trends for Influent and Effluent TSS for Seven HSSF Wetland Systems in England

System Location	Waste Stream	C_{mean} (mg/L)	Amplitude, A	t_{max} (Julian day)	R^2
Tertiary Systems					
Leicestershire	Influent	19	0.26	361	0.07
	Effluent	4	0.15	338	0.02
Solihull	Influent	19	0.24	124	0.11
	Effluent	5	0.32	165	0.12
Staffordshire	Influent	41	0.23	237	0.04
	Effluent	6	0.34	198	0.02
Secondary Systems					
Yorkshire	Influent	162	0.10	115	0.09
	Effluent	24	0.13	202	0.06
North Yorkshire 1	Influent	106	0.21	197	0.01
	Effluent	35	0.21	215	0.19
North Yorkshire 2	Influent	144	0.04	261	0.01
	Effluent	32	0.35	119	0.20
North Yorkshire 3	Influent	144	0.04	261	—
	Effluent	34	0.32	112	0.05

Note: Site names for U.K. systems are approximate. The trend in each time series is presumed to be sinusoidal (see text for notation). $C = C_{mean}(1 + A\cos[\omega(t - t_{max})]) + E$

Source: CWA database (2006) *Constructed Wetlands Interactive Database, Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.

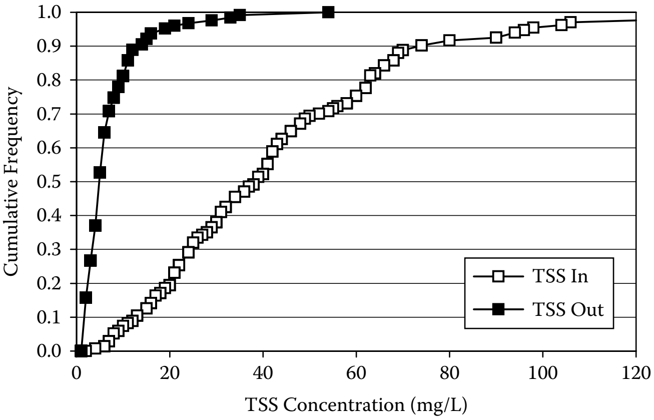


FIGURE 7.29 Frequency percentiles for inlet and outlet TSS for a HSSF wetland in Staffordshire, England. The median inlet TSS was 38 mg/L; the median outlet TSS was 5 mg/L. Four years of data are represented; sample frequency was every two weeks. (Data from CWA database (2006) *Constructed Wetlands Interactive Database, Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

7.6 TSS REMOVAL IN VF WETLANDS

VF wetlands span a wide variety of operating regimes. To ensure adequate influent distribution, VF wetlands are often operated on a pulse-load (surface flooding) regime, tidal-flow (fill-and-drain) regime, or with networks of perforated distribution pipes. The fate, transport, and generation of TSS in vertical flow wetlands depends on several factors, including:

- Flow mode (continuous or intermittent; saturated or unsaturated)
- Inlet organic loading (particulate + soluble)
- Loading of inert (mineral TSS)
- Size of the wetland bed media

The wide range of flow modes and media sizes presents considerable difficulty when generalizing TSS fate, generation and transport regarding VF wetlands. Instead, this section focuses on the most common VF wetland mode, intermittent downflow beds. Other types of VF wetland systems are discussed in more detail in Part II of this book.

INTERMITTENT DOWNFLOW BEDS

As discussed in Chapters 1 and 2, these types of VF wetlands are pulse-loaded, and operate on principles of unsaturated flow. Typically, these systems use a relatively fine, sand bed media ($d_{10} \sim 0.25$ mm) (Gesellschaft zur Förderung der Abwassertechnik d.V (GFA), 1998; Brix and Johansen,

TABLE 7.10
Trend Multipliers for Effluent TSS in Tertiary (N = 3) and Secondary (N = 4) HSSF Wetlands in England

			Excursion Frequency		
Location	**Median Inlet (mg/L)**	**Median Outlet (mg/L)**	**80%**	**90%**	**95%**
Leicestershire	14	4	1.30	2.00	2.50
Solihull	18	4	1.75	2.00	2.25
Staffordshire	38	4	2.25	3.10	4.45
Mean for Tertiary Systems			**1.77**	**2.37**	**3.07**
Yorkshire	154	22	1.35	1.81	2.39
North Yorkshire 1	100	34	1.55	1.91	2.10
North Yorkshire 2	144	29	1.50	1.78	2.05
North Yorkshire 3	144	30	1.47	1.85	2.25
Mean for Secondary Systems			**1.47**	**1.84**	**2.20**

Note: Site names for U.K. systems are approximate.

Source: CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom. Trend multiplier is $(1 + \Psi)$; see Equation 6.61.

2006). With this bed media, deposition and filtration of particulates, especially near the surface of the bed, is a dominant removal mechanism of incoming TSS (Woodward and Ta, 1988; Platzer and Mauch, 1997).

In addition to the accumulation of incoming TSS at or near the surface of the wetland, a surficial biomat may develop on top of the VF bed in response to the overall organic loading (Figure 7.31). The mechanisms of biomat formation are essentially similar to those of biosolids accumulation encountered internally in HSSF wetlands. However, there is one important difference in the biomat formation in HSSF and VF wetlands. In HSSF wetlands, the loading is typically continuous, in a saturated flow environment. In VF wetlands, the loading is intermittent, allowing for "resting" periods of no biomat formation. For VF wetlands that are unsaturated during the resting phase, the high availability of atmospheric oxygen (21%) aids in aerobic decomposition of accumulated biomat material. Further,

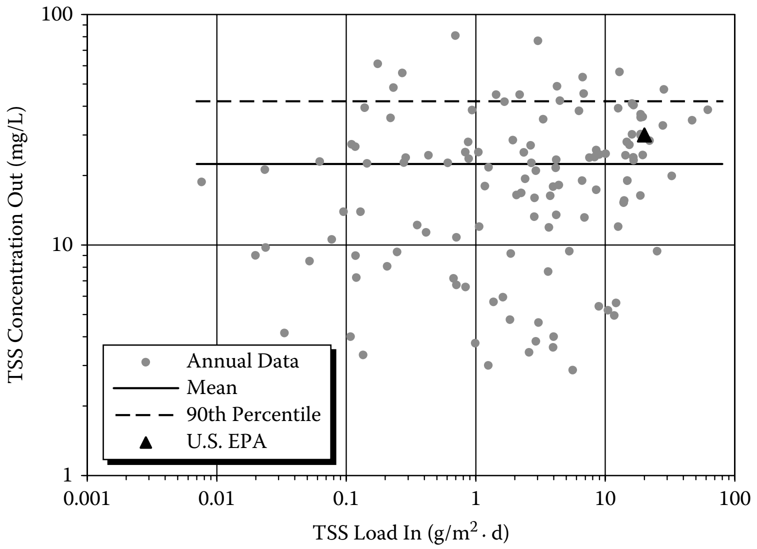


FIGURE 7.30 TSS in the effluent of HSSF wetlands is not a function of inlet TSS loading. Chart represents 26 systems and 130 system-years of performance data from the CWA (2006) and WERF (2006) databases. The mean is 22.5 mg/L and the 90th percentile is 42 mg/L.

FIGURE 7.31 Accumulated particulate organic matter on top of an intermittent downflow VF wetland bed in Roussillon, France.

the material is more accessible, because of its top location. Because of this configuration, VF wetlands are much more amenable to a "load and rest" operational regime to mitigate TSS clogging.

Input–Output Relations

Removal of TSS is primarily through the mechanisms of filtration and interception, as is readily seen in Figure 7.31. This is broadly related to TSS loading, as indicated in Figure 7.32, load-in–concentration-out plot of intermittent downflow beds.

Vertical flow wetlands are highly effective in TSS removal, provided they are managed in a way to avoid bed clogging problems (typically accomplished through a load and rest operation regime). Data presented in Figure 7.32 represent 71 system-years of data from 31 vertical flow wetlands (intermittent downflow beds). The median inlet TSS concentration (C_i) was 90 mg/L; the median outlet TSS concentration (C_o) was 12 mg/L (87% concentration reduction).

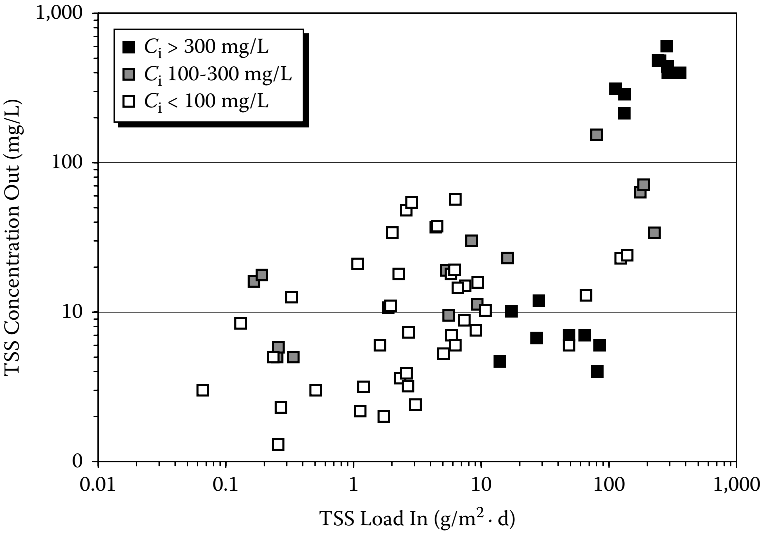


FIGURE 7.32 Inlet TSS loading versus effluent TSS concentrations for VF wetlands. Data show 71 system-years of data from 31 intermittent downflow wetlands.

SUMMARY

Treatment wetlands are consistently effective at reducing elevated concentrations of suspended solids. As most treatment wetlands are overdesigned in terms of TSS reduction, existing data are primarily useful for estimating background TSS outflow concentration variability, and are not helpful in estimating an area-based reaction rate constant for this parameter. If there is a need to specifically target TSS removal with more accuracy, procedures should rely on measured settling rates for the specific wastewater, while taking due caution to recognize the inevitable internal wetland processes that will result in irreducible background TSS concentrations and stochastic variability in response to factors outside the treatment wetland operator's control.

Suspended solids removal in FWS wetlands occurs through sedimentation, aggregation, and filtration/interception. FWS wetlands are generally very efficient in removing suspended solids, but foraging wildlife and wind- and wave-induced mixing can resuspend particulate matter. Phytoplankton production can also increase the concentration of suspended sediments in FWS wetlands. Suspended solids in HSSF and VF wetland systems generally do not exhibit particle resuspension because wind- and animal-induced mixing of the water column does not occur. Most suspended solids in HSSF and VF wetlands are removed through sedimentation and filtration or interception. High influent loading of suspended solids can lead to excessive biological growth which may clog sand or gravel media, causing head loss through the system that may lead to overland flow in HSSF wetlands and a complete failure of VF systems. Design approaches and operational responses to bed clogging in SSF wetlands are discussed in Part II of this book.

8 Carbon and Biochemical Oxygen Demand

Carbon compounds interact strongly with wetland ecosystems. The carbon cycle in wetlands is vigorous and typically provides carbon exports from the wetland to receiving ecosystems. Many internal wetland processes are fueled by carbon imports and by the carbon formed from decomposition processes.

Treatment wetlands frequently receive large external supplies of carbon in the added wastewater. Any of several measures of carbon content may be made, with biochemical oxygen demand (BOD) being the most frequent in the treatment of municipal wastewater. Degradable carbon compounds are rapidly utilized in wetland carbon processes. At the same time, a variety of wetland decomposition processes produce available carbon. The balance between uptake and production provides the carbon exports. In general, the amounts of carbon cycled in the wetland are comparable to the quantities added in domestic wastewater.

The growth of wetland plants requires carbon dioxide (CO_2) for photosynthesis. A variety of organisms release CO_2 as a product of respiration. Many pathways lead to the microbial production of CO_2, as well as methane (CH_4). Both gases dissolve in water to a limited extent; so there are active transfers of carbon to and from the atmosphere.

In terms of treatment, it is therefore not surprising to find good carbon reductions for the added wastewater, accompanied by nonzero background levels of various carbon compounds and the related BOD. For purposes of wetland design for BOD removal, the challenge is to find relatively simple design tools despite the enormously complex set of wetland functions.

8.1 WETLAND CARBON SPECIATION AND PROCESSING

A wide spectrum of carbon compounds exists in either dissolved or particulate forms in aquatic systems. The usual dividing line is a 0.45-μm filter. The following distinctions are made as a result of analytical methods:

- TC = total carbon (includes all dissolved and suspended forms)
- PC = particulate carbon (includes organic and inorganic forms)
- DC = dissolved carbon (includes organic and inorganic forms)
- IC = inorganic carbon (includes all dissolved and suspended forms)
- DIC = dissolved inorganic carbon (usually comprises CO_2, carbonate, and bicarbonate)
- TOC = total organic carbon (includes all dissolved and suspended forms)
- DOC = dissolved organic carbon
- NDOC = nondissolved organic carbon
- VOC = volatile organic carbon (compounds)

In soils or biomass, samples are subjected to combustion and dissolution, followed by analysis for total carbon.

BOD, COD, AND TOC

Different analytical techniques are used to measure the amount of organic material in the wastewater. BOD is a measure of the oxygen consumption of microorganisms in the oxidation of organic matter. It is measured as the oxygen consumption in an airtight incubation of the sample. This test normally runs for five days, and the result is then more properly designated as BOD_5. Some oxygen may be used in nitrification if the necessary organisms are present in the sample. If this potential nitrogenous oxygen demand is inhibited chemically during the test, the result is carbonaceous biochemical oxygen demand ($CBOD_5$).

Chemical oxygen demand (COD) is the amount of a chemical oxidant, usually potassium dichromate, required to oxidize the organic matter. This measure is larger than BOD, because the strong oxidant attacks a larger group of compounds. However, nitrogenous compounds, such as ammonia, are not oxidized by the COD test. Oxygen or oxidant consumption may be measured before or after filtration, leading to measures of total and soluble BOD and COD. In the wetland environment, the presence of humic materials leads to COD values that are much larger than BOD values. In a northern peatland, the ratio was approximately 0.05 (BOD_5 = 5 mg/L:COD = 100 mg/L) (unpublished data from the Houghton Lake, Michigan, peatland). At Tres Rios, Arizona, wetlands treating nitrified secondary effluent, four wetlands gave ratios of 0.055 ± 0.004, averaged over seven years. In municipal wastewaters, the ratio is typically 0.4–0.8 (Metcalf and Eddy, Inc., 1991). Industrial wastewaters may have lower ratios.

Total organic carbon (TOC) is measured by chemical oxidation followed by analysis for CO_2. In a northern peatland, the ratio BOD_5:TOC was approximately 0.2 (BOD_5 = 5 mg/L:TOC = 25 mg/L) (unpublished data from the Houghton Lake peatland), and was 0.28 at Estevan, Saskatchewan. At Tres Rios wetlands treating nitrified secondary effluent, four wetlands gave ratios of $CBOD_5$:TOC = 0.25 ± 0.08, averaged over

TABLE 8.1
Comparison of Oxygen Consumption Parameters for Various Waters

	BOD_5/COD	$CBOD_5$/COD	BOD_5/TOC	$CBOD_5$/TOC
From Crites and Tchobanoglous				
Untreated wastewater	0.3–0.8	—	1.2–2.0	—
After primary settling	0.4–0.6	—	0.8–1.2	—
Final effluent	—	0.1–0.3	—	0.2–0.5
From Metcalf and Eddy				
Untreated wastewater	0.4–0.8	—	1.0–1.6	—
FWS Wetland Effluents				
Columbia, Missouri	0.21–0.23	0.11–0.13	—	—
Tres Rios, Arizona	0.05–0.06	—	—	0.17–0.33
Estevan, Saskatchewan	—	—	0.28	—
Houghton Lake, Michigan	0.05	—	0.2	—
Orlando Easterly, Florida	—	—	0.09–0.13	—

Source: WWTP values from Crites and Tchobanoglous (1998) *Small and Decentralized Wastewater Management Systems.* McGraw-Hill, New York; Metcalf and Eddy Inc. (1991) *Wastewater Engineering, Treatment, Disposal, and Reuse.* Tchobanoglous and Burton (Eds.), Third Edition, McGraw-Hill, New York.

seven years. In municipal wastewaters, the ratio is 1.0:1.6 (Metcalf and Eddy Inc., 1991).

The interrelation among the various measures of carbon and oxygen demand are given in Table 8.1. The interpretation of these ratios is that natural wetlands cycle at low levels of biologically usable carbon compounds, whereas municipal wastewaters are rich in usable carbon compounds.

Wetlands are efficient users of external carbon sources, manifested by excellent reductions in BOD_5 and COD. However, wetlands possess nonzero background levels of both BOD and COD, which depend on the type and status of the wetland. Typical ranges for background concentrations are 1–10 mg/L for BOD_5 and 10–100 mg/L for COD.

Wetland Chemistry of Carbon

Dissolved Inorganic Carbon

Of the hundreds of carbon compounds that may occur in the wetland environment, relatively few are inorganic. Dissolved inorganic carbon consists primarily of CO_2, carbonate, and bicarbonate.

In pure water solution, the principal carbonate species are related to atmospheric CO_2 by the temperature and pH-dependent dissolution and dissociation series:

Henry's Law:

$$H_2CO_3^* \rightleftharpoons H_2O + CO_{2(g)} \qquad K_H = \frac{P_{CO_2}}{[H_2CO_3^*]} \tag{8.1}$$

where

$$[H_2CO_3^*] = [H_2CO_3] + [CO_2] \tag{8.2}$$

Hydration:

$$H_2CO_3 \rightleftharpoons H_2O + CO_2 \qquad K = \frac{[CO_2]}{[H_2CO_3]} \tag{8.3}$$

First Dissociation:

$$H_2CO_3 \rightleftharpoons HCO_3^- + H^+ \qquad K_{H_2CO_3} = \frac{[HCO_3^-][H^+]}{[H_2CO_3]} \tag{8.4}$$

Second Dissociation:

$$HCO_3^- \rightleftharpoons CO_3^{2-} + H^+ \qquad K_2 = \frac{\left[CO_3^{2-}\right][H^+]}{[HCO_3^-]} \tag{8.5}$$

and where, as a result of Equation 8.2,

$$K_1 = \frac{K_{H_2CO_3}}{K+1} \tag{8.6}$$

the notation of Pankow (1991) has been adopted. Brackets indicate the concentration of the chemical species, in molarity; and all are in water except for atmospheric CO_2. The value of the equilibrium constant $K \approx 650$, and hence most of the dissolved carbon is present as CO_2. Equations 8.1–8.6 may be solved for concentrations, given the partial pressure of CO_2 and the various equilibrium constants.

$$[H_2CO_3^*] = K_H\, P_{CO_2} \tag{8.7}$$

$$[HCO_3^-] = \frac{K_1}{[H^+]} K_H\, P_{CO_2} \tag{8.8}$$

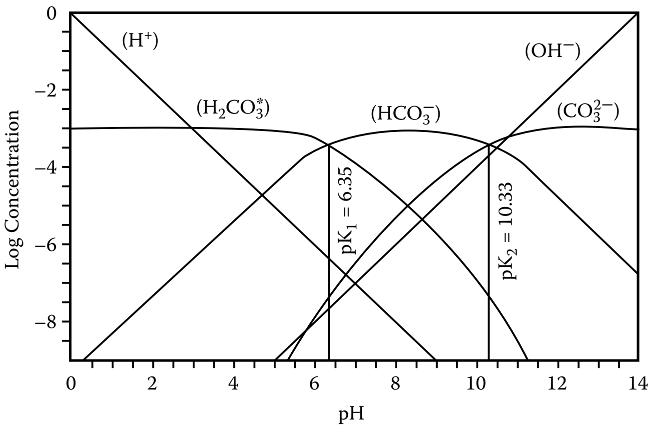


FIGURE 8.1 Distribution of carbonate species in water at 25°C. The partial pressure of CO_2 in the air is taken as 3.16×10^{-4} atm. (From Metcalf and Eddy Inc. (1998) *Wastewater Engineering, Treatment, Disposal, and Reuse,* Tchobanoglous *et al.* (Eds.), Fourth Edition, McGraw-Hill, New York. Reprinted with permission.)

$$\left[CO_3^{2-}\right] = \frac{K_1 K_2}{[H^+]^2} K_H \, P_{CO_2} \tag{8.9}$$

The equilibrium constants, and hence the various concentrations, are all pH- and temperature-dependent. These forms are distributed in water at 25°C as shown in Figure 8.1 (Pankow, 1991). However, it must be noted that wetland waters are more complex than the pure water system and therefore will not follow such idealized chemistry precisely. Modifications of the calculation (APHA, 1992) deal with expected deviations due to dissolved solids, but not the full suite of biological variations that may be expected in wetlands. Production and consumption of CO_2 in the wetland may significantly alter the chemical balance in the water.

An important feature of the carbonate system is its influence on pH under mediation by algae. Algal consumption of CO_2 drives pH upward, and may give rise to $9 < pH < 10$ in unshaded wetland environments or ponds.

Precipitates

A variety of cations can precipitate carbonates under certain conditions. The most important is calcium carbonate, $CaCO_3$. A major process in periphyton-dominated wetlands is chemical precipitation of $CaCO_3$ under conditions of high pH created by the algae (Gleason, 1972). Similarly, in beds of submerged aquatic vegetation, CO_2 and bicarbonate are consumed during photosynthesis, thereby raising the water column pH and promoting $CaCO_3$ precipitation (Dierberg *et al.*, 2002).

A variety of cations can precipitate carbonate under certain conditions. Some important mineral precipitates in the wetland environment are:

Calcite:	$CaCO_3$
Aragonite:	$CaCO_3$
Magnesite:	$MgCO_3$
Dolomite:	$CaMg(CO_3)_2$

Calcium carbonate saturation indices may be calculated in a number of ways (APHA, 1992). However, overall carbon mineral chemistry is very complex; consequently, accurate calculations of solubilities are generally not possible, especially in wetland environments.

Organic Carbon

Biomass: Growth, Death, Decomposition

The wetland cycle of growth, death, and partial decomposition uses atmospheric carbon, and produces gases, dissolved organics, and solids (Figure 8.2). Decomposition involves the sugars, starches, and low molecular weight celluloses in the dead plant material. Gaseous products include methane and regenerated CO_2. A spectrum of soluble large organic molecules, collectively termed *humic substances*, are released into the water. The solid residual of plant decomposition is peat or organic sediment, which originated as celluloses and lignins in the plants. These wetland soil organics are broadly classified as fulvic material, humic material, and humin, based upon whether they are acid soluble, base soluble, or insoluble (NRCC, 1979).

The sediments, soils, and biomass in a wetland contain major proportions of carbon. The carbon content of 28 species of wetland plants has been reported by Boyd (1978) as 41.1% ± 0.7% (dry weight, mean ± SE). *Typha latifolia* values from 30 sites ranged from 43.3% to 47.2% (Boyd and Hess, 1970). Reddy *et al.* (1991) reported 44.0% ± 2.5% for peats in the upper 30 cm of the soil column. Soil scientists sometimes use a concentration of 58% for the carbon content of soil organic matter (the Van Bemmelen factor; Collins and Kuehl, 2001). Thus nearly half of the dry wetland plant and soil material is carbon.

The internal wetland carbon cycle is large. A general idea of the magnitudes of the various carbon transfers in a northern treatment marsh may be gained from considering the annual growth and decomposition patterns (see Chapter 3). A eutrophic treatment marsh grows about 3,000 dry g/m^2 of aboveground biomass each year, with a carbon content of about 43%. This translates to an annual average requirement for 35 kg/ha·d of carbon. In northern climates, this requirement is utilized during a growing season of approximately four months. In the case of emergent macrophytes, some of this carbon may be withdrawn from the atmosphere. However, submerged vegetation draws carbon from the aquatic carbonate system.

Decomposition of the resultant litter returns a significant portion of that carbon to the atmosphere and to wetland waters, but in treatment wetlands, a small fraction, on the order of 15% or 20%, is stored in accreted soil and sediments. That storage (burial) fraction therefore amounts to about 5 kg/ha·d as an annual average for the eutrophic marsh example. The balance, about 30 kg/ha·d, is processed via one or more mechanisms involving a variety of electron acceptors (oxidants), or via anaerobic digestion which generates methane.

The oxygen consumed by aerobic decomposition of sediments and litter is termed the sediment oxygen demand (SOD). In stream environments with large wastewater influences, the rate of consumption of oxygen by the stream sediments may be estimated as 20–100 kg/ha·d (Metcalf and

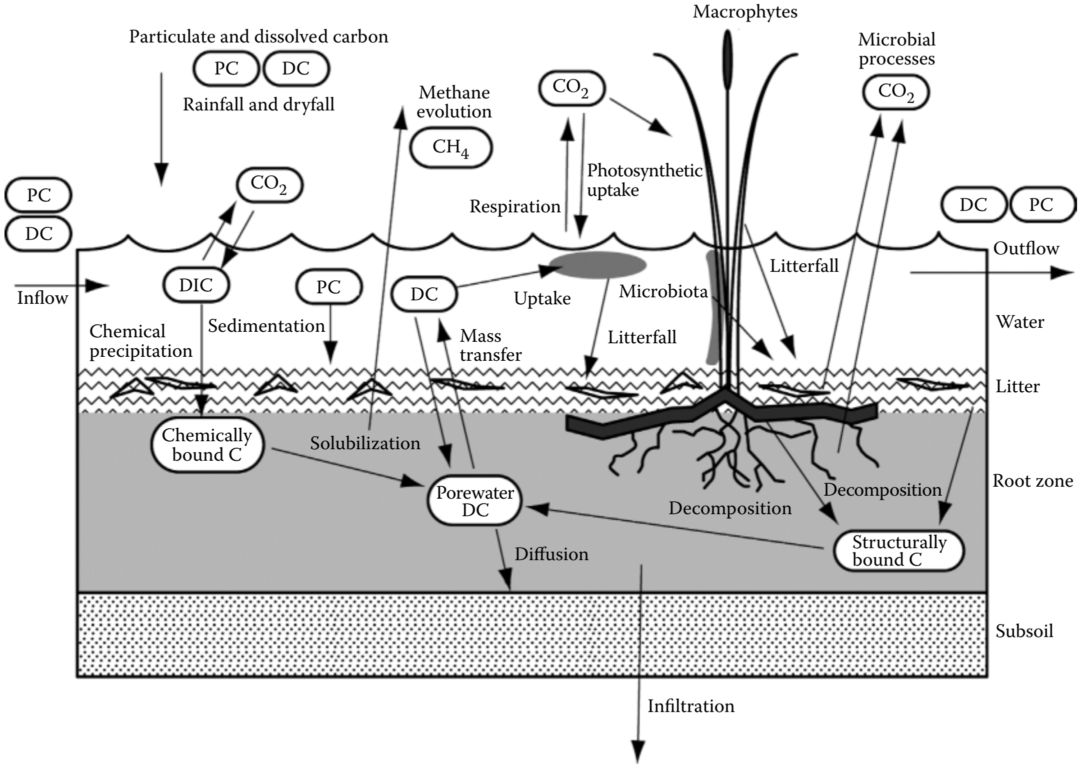


FIGURE 8.2 Carbon storages and transfers in the wetland environment. DC = dissolved carbon; PC = particulate carbon; DIC = dissolved inorganic carbon; DOC = dissolved organic carbon; CH_4 = methane; CO_2 = carbon dioxide. Biomass carbon consists of living and dead biomass, as well as organic decomposition products. (From Kadlec and Knight (1996) *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

Eddy Inc., 1991). In the eutrophic marsh example, if all the decomposition were to proceed via oxidation with dissolved oxygen as the electron acceptor, and CO_2 as the product, the equivalent SOD loading would be $(32/12) \times 30 = 80$ kg/ha·d. As will be subsequently shown, this potential SOD loading is at the upper end of the range of external BOD loadings (BLI) for treatment wetlands.

The wetland environment is more complicated than the stream environment. Some of the carbon is processed above-water, as standing dead material oxidizes. Some of the submerged sediments and litter are processed into soluble organic compounds that contribute to CBOD in the water, thus creating a nonzero background CBOD in a wetland environment. Starches, sugars, and cellulose are degraded to amino acids and fatty acids (Reddy and Graetz, 1988). In addition to dissolved oxygen, a variety of electron acceptors may be involved in decomposition.

Carbon Processing in Wetland Necromass and Soils

A rough representation of the various decomposition "reactions" may be set down (Mitsch and Gosselink, 1993). These occur in different horizons in the wetland, as indicated in Figure 8.3.

Respiration occurs in aerobic zones:

$$\underset{\text{carbohydrates}}{C_6H_{12}O_6} + 6\,O_2 \rightarrow 6\,CO_2 + 6\,H_2O \qquad (8.10)$$

Fermentation occurs in anoxic or anaerobic zones:

$$\underset{\text{carbohydrates}}{C_6H_{12}O_6} \rightarrow 2\,\underset{\text{lactic acid}}{CH_3CHOHCOOH} \qquad (8.11)$$

$$\underset{\text{carbohydrates}}{C_6H_{12}O_6} \rightarrow 2\,\underset{\text{ethanol}}{CH_3CH_2OH} + 2\,CO_2 \qquad (8.12)$$

Nitrate Reduction (denitrification) occurs in anoxic or anaerobic zones:

$$\underset{\text{carbohydrates}}{C_6H_{12}O_6} + 4\,NO_3^- \rightarrow 6\,CO_2 + 6\,H_2O + 2\,N_2 + 4\,e^- \qquad (8.13)$$

Iron Reduction occurs in anoxic or anaerobic zones:

$$\underset{\text{acetate}}{CH_3COO^-} + 8\,Fe^{3+} + 3\,H_2O \rightarrow 8\,Fe^{2+} + CO_2 + HCO_3^- + 2\,H_2O + 8\,H^+ \qquad (8.14)$$

Sulfate Reduction occurs in anaerobic zones:

$$2\,\underset{\text{lactate}}{CH_3CHOHCOO^-} + SO_4^{2-} + H^+ \rightarrow 2\,\underset{\text{acetate}}{CH_3COO^-} + 2\,CO_2 + 2\,H_2O + HS^- \qquad (8.15)$$

$$\underset{\text{acetate}}{CH_3COO^-} + SO_4^{2-} + 2\,H^+ \rightarrow 2\,CO_2 + 2\,H_2O + HS^- \qquad (8.16)$$

Methanogenesis occurs in anaerobic zones:

$$4H_2 + CO_2 \rightarrow CH_4 + 2\,H_2O \qquad (8.17)$$

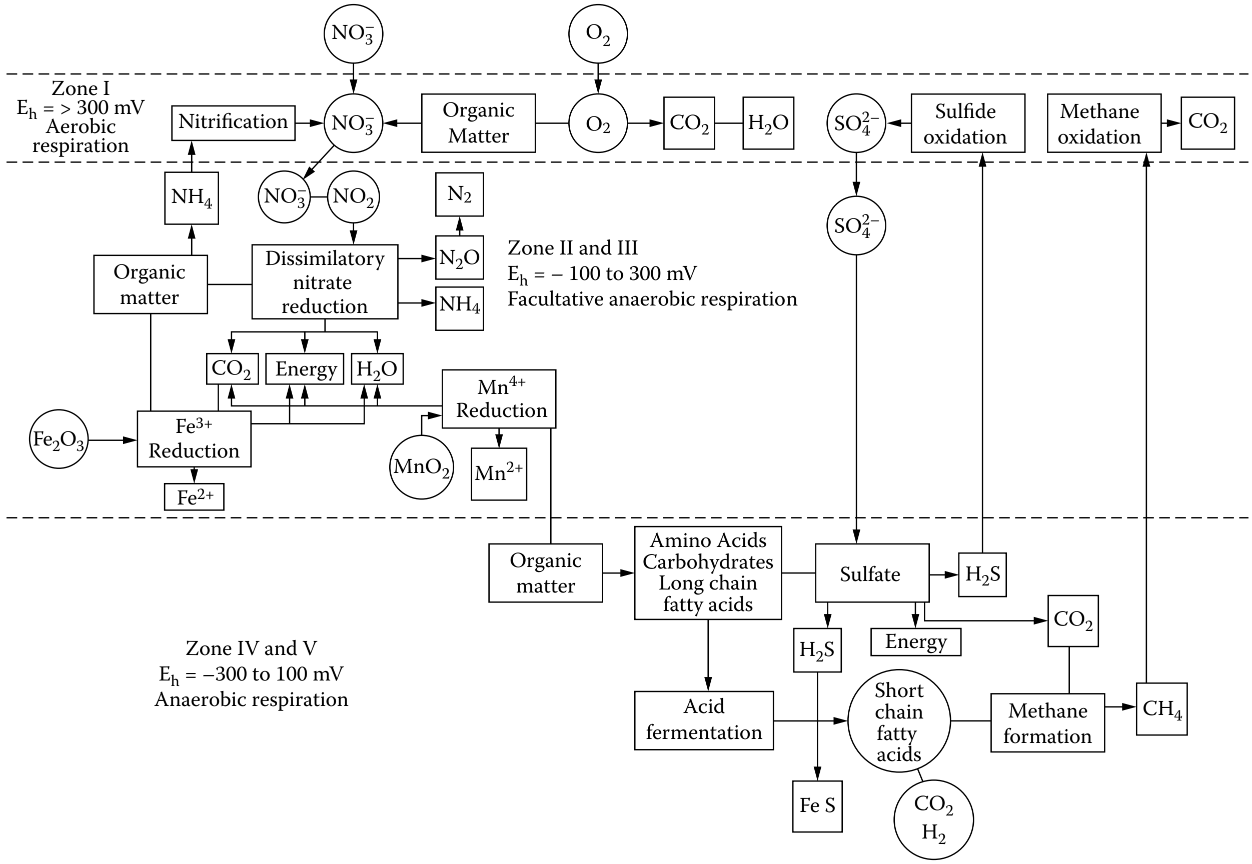


FIGURE 8.3 Pathways of organic carbon decomposition in wetland soils. Aerobic, facultative anaerobic, and obligate anaerobic processes are all typically present at different depths in the soil. (From Reddy and Graetz (1988) In *The Ecology and Management of Wetlands*. Hook (Ed.), Croom Helm, London, United Kingdom, pp. 307–318. Reprinted with permission.)

$$\underset{acetate}{CH_3COO^-} + 4\,H_2 \rightarrow 2\,CH_4 + H_2O + OH^- \tag{8.18}$$

The relative percentages of these reactions were investigated in controlled SSF wetland microcosms by Burgoon (1993), using acetate as the carbon source. His results demonstrated that all routes can be important, depending upon physical and chemical conditions.

It is apparent that the wetland provides a spectrum of potential pathways for the utilization of organic carbon compounds. Sufficient information is not available to quantify both the complex chemistry and the spatial distribution of chemical compounds. Therefore, the interactions must be described via correlations and rate equations, which are supportable by wetland performance data.

8.2 BOD REMOVAL IN FWS WETLANDS

A large amount of BOD data now exists for FWS wetlands treating a variety of wastewaters. There are a number of ways to summarize this information, including removal rate models and graphical summaries. When waters with moderate to large concentrations of BOD flow through a wetland, a decrease in concentration to a nonzero plateau is typically observed. This behavior is illustrated in Figure 8.4 for one of the continuous flow Sacramento, California, wetlands (Nolte and Associates, 1997). Samples were taken along the wetland

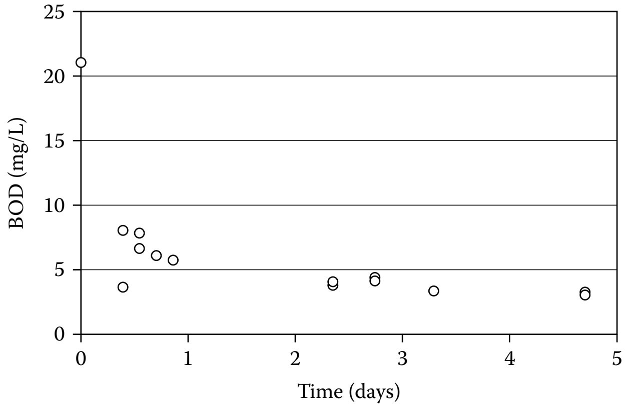


FIGURE 8.4 Profiles of BOD concentration in Cell 7B of the Sacramento, California, treatment wetlands on May 3 and May 4, 1995. The plateau is at 3.1 mg/L. (Data from Nolte and Associates (1997) *Sacramento Regional Wastewater Treatment Plant Demonstration Wetlands Project*. 1996 Annual Report to Sacramento Regional County Sanitation District, Nolte and Associates: Sacramento, California.)

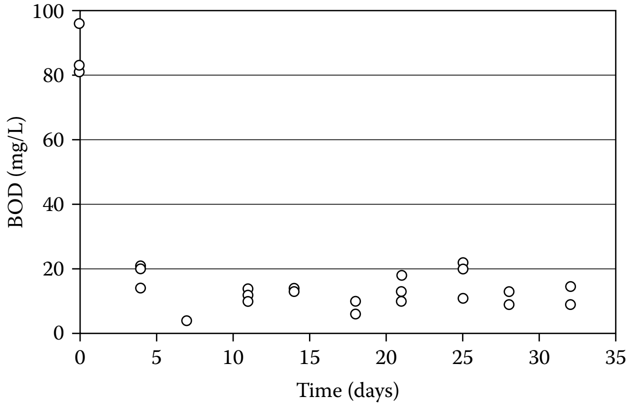


FIGURE 8.5 The progression of BOD concentrations in three wetlands operated in the batch mode. The plateau is at 11.3 mg/L (Data from Lakhsman (1981) *A Demonstration Project at Humboldt to Provide Tertiary Treatment to the Municipal Effluent Using Aquatic Plants*. SRC Publication No. E-820-4-E-81. 74 pp. Saskatchewan Research Council.)

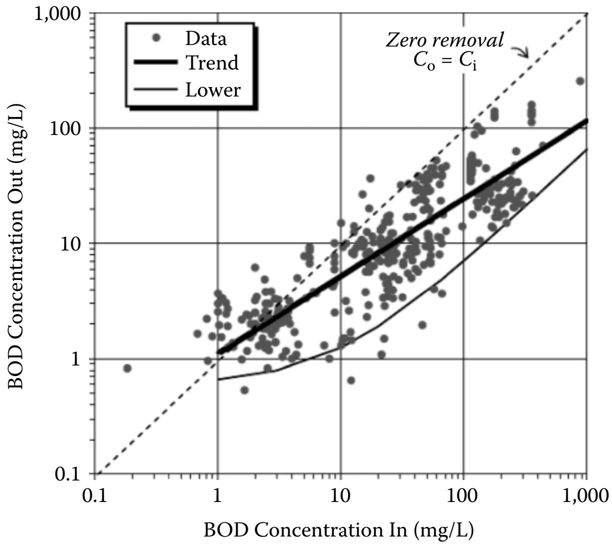


FIGURE 8.6 Input–output concentration for BOD in FWS wetlands. Each point represents an annual average for one wetland. There are 385 wetland·years of data for 138 wetlands. The trend line is $y = 1.13x^{0.67}$ ($R^2 = 0.75$ logarithmic). The lower bound line is $y = 0.6 + 0.065\,x$, and includes 98% of the annual averages.

length, at positions corresponding to increasing nominal detention time. The same sort of response is seen in the results of Lakhsman (1981) for batch wetland treatment of lagoon effluents. A set of wetlands were charged with wastewater, then closed in, with no water additions or withdrawals. Typical response data showed a sharp decrease in BOD_5 to a nonzero, fluctuating background (Figure 8.5). The decrease is steep—perhaps exponential—but to a nonzero background BOD_5.

Annual Input–Output Concentration Relations

The concentration of carbonaceous compounds is reduced in FWS wetlands for incoming concentrations above background. If, however, incoming BOD is below background, concentrations may increase upon passage through the system. As inlet concentrations increase, outlet concentrations increase, in a log-linear progression (Figure 8.6). There is considerable intersystem variability, but the data exhibit a lower bound, which may be interpreted as the lowest background concentration corresponding to a given inlet concentration. This curve is approximated by

$$C^{**} = 0.6 + 0.065\,C_i \tag{8.19}$$

where

C_i = inlet BOD concentration, mg/L
C^{**} = lower limit background BOD concentration, mg/L

Depending on hydraulic conditions, and the character of the incoming BOD, individual wetlands will typically exhibit different C^*-values as model calibration parameters, which may be larger than C^{**}.

First-Order Modeling

The *P-k-C** first-order model can readily account for observations, for appropriate values of parameters (see Chapter 6). However, parameter values are known to depend on system hydraulics (Kadlec, 2000), as well as on speciation of the BOD (Crites and Tchobanoglous, 1998; Kadlec, 2003a).

BOD and COD are water quality parameters measured by procedures that lump individual chemical compounds into an overall, or total, concentration for that class of materials. It is clear that the individual components of such mixtures may be degraded or removed at different rates, and that there is a corresponding difference in removal rate constants (Crites and Tchobanoglous, 1998; Tchobanoglous *et al*., 2000; Kadlec, 2003a). There is therefore a distribution of rate constants across the various mass fractions of the mixture. As water containing such a mixture passes through the wetland, its composition changes because different fractions of the mixture are reduced at different rates. The mixture becomes *weathered*, a term coined to describe the selective stripping of light volatile materials upon exposure to outdoor environments. In the case of BOD and COD, the easy-to-degrade substances are lost first; more recalcitrant compounds persist for longer times.

The BOD test itself reflects only a fraction of the carbonaceous mixture, because it is terminated before all components are oxidized. For municipal wastewater, the five-day BOD test typically measures about two thirds of the ultimate BOD (UOD) (Metcalf and Eddy, Inc., 1991; Crites and Tchobanoglous, 1998).

Effects of Lumping on Removal Models

The potential effects of speciation in lumped contaminant measures, particularly BOD, as manifested in changing rates, have been known for several years (Tchobanoglous, 1969; Crites and Tchobanoglous, 1998; Shepherd *et al*., 2001).

Crites and Tchobanoglous (1998) set forth a formulation for a "retarded rate expression." However, Kadlec (2003a) demonstrated that this concept was subsumed by a relaxed tanks-in-series (TIS) model. The *P-k-C** model is here defined to be (see Chapter 6):

$$\frac{C_o - C^*}{C_i - C^*} = \frac{1}{(1 + k/Pq)^P} \tag{8.20}$$

where
- C_i = inlet BOD concentration, mg/L
- C_o = outlet BOD concentration, mg/L
- C^* = background BOD concentration, mg/L
- k = apparent TIS rate constant, m/yr
- P = apparent number of TIS for BOD reduction
- q = hydraulic loading rate, m/yr

The parameter P accounts for two effects: the detention time distribution (DTD) and the k-value distribution (kVD) (see Chapter 6). The value of P is always less than the number of tanks determined from a tracer test. For broad distributions of k-values, such as may occur for BOD, a hydraulic TIS number of four (see Table 6.3) will be reduced to a P-value of one or two. However, the C^*-value in Equation 8.20 reflects several possible different causes. There may be a real irreducible component of BOD (hard to imagine, because it all disappears in the lab test), or there may be wetland ecosystem feedback of BOD constituents. But in addition, DTDs and kVDs may create an apparent C^* as an artifact of model parameter fitting. These may be considered "bypassing C^*" and "weathering C^*", respectively.

Reasonable data fits may be obtained for specific wetlands or specific sites. Seven Gustine, California, wetlands were operated at different hydraulic loadings (different detention times) for a calendar year (Walker and Walker, 1990). The *P-k-C** model parameters determined from that input–output data were: $P = 1$, $k = 63$ m/yr, and $C^* = 9.7$ mg/L ($R^2 = 0.60$). Those parameters also provided a reasonable fit to transect data (Figure 8.7, $R^2 = 0.59$). However, it is uncommon to have multiple wetlands and multiple loadings from which to derive these types of calibrations.

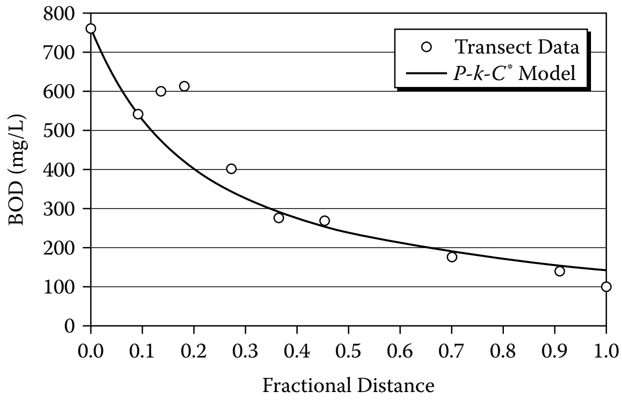


FIGURE 8.7 BOD profile in the flow direction for wetland 1D at Gustine, California. The model curve was derived from independent input–output data for seven wetlands over a calendar year. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

Concentration Profiles and Modeling Pitfalls

Difficulties with the *P-k-C** first-order model are compounded by the problem that data sets are very often poorly conditioned to produce good estimates of both k and C^* by any of the several methods of parameter estimation. This is easily visualized from Figures 8.4, 8.5, and 8.8, which contain examples of the early exponential decline (governed by k), together with the late plateau (governed by C^*). There are insufficient data in the exponential region for Sacramento and Humboldt to get a good estimate of k, but plenty of data to define C^*. Conversely, the Arcata pilot, Benton, and Gustine data sets never reach a plateau; all the data is concentrated in the exponential decline region. Thus, for these wetlands, transect data will provide a good estimate of k, but a very poor estimate of C^*. Input–output data for these sites may nonetheless be fitted to the model. In addition to the Gustine results given above, Benton input–output data over a two-year span resulted in $P = 1$, $k = 260$ m/yr, and $C^* = 5$ mg/L. At the Arcata pilot, input–output data over a two-year span resulted in $P = 1$, $k = 53$ m/yr, and $C^* = 4$ mg/L.

It is tempting to arbitrarily pick some low concentration to represent C^*, but that is counter-indicated by the importance of C^* in wetland sizing, as shall be seen in the following sections. There is not an existing method to make such an estimate with confidence. One need look no further than data from two wetlands in the same geographical region: Humboldt, Saskatchewan, shows $C^* = 11.3$, but not far away, Oak Hammock, Manitoba, shows $C^* = 2.4$. Both are batch systems treating domestic lagoon effluent. We shall also see that k-values are widely variable, both across years for one wetland (interannual variability) and across wetlands (intersystem variability). Thus, to the dismay of researchers seeking to do THE definitive design model calibration study, no such study can be trusted in and of itself.

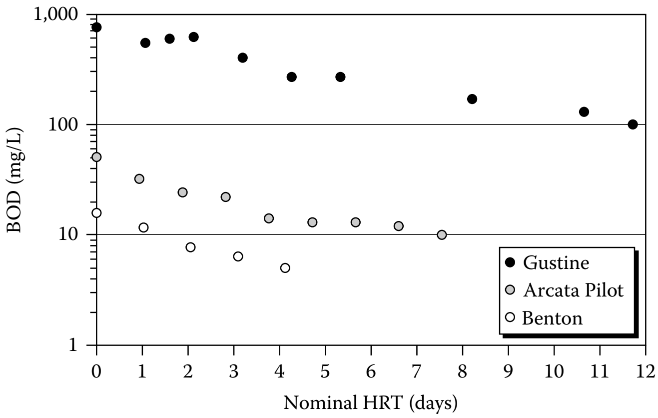


FIGURE 8.8 Initial exponential declines in BOD for FWS wetlands. These systems did not achieve any apparent plateau.

Distribution of *k*-Values

It is instructive to examine multiple data sets that provide a distribution of *k*-values and C^*-values. If all data are considered together, the inter- and intrasystem effects are compounded by a shift in the probable mechanisms of BOD reduction, as detailed in Equations 8.10–8.18. As loadings increase, aerobic processes become less of a probable factor, and are replaced by anoxic processes. Therefore, four levels of inlet concentration are considered: tertiary ($0 < C_i < 30$ mg/L); secondary ($30 < C_i < 100$ mg/L); primary ($100 < C_i < 200$ mg/L); and "super" ($C_i > 200$ mg/L). The effect of BOD weathering, which produces lower *k*-values as reaction proceeds, is quite strong for BOD. Data fits are better for *P*-values that are considerably lower than the tracer-determined number of tanks-in-series (*N*TIS) values. In general, data fits are best at $P = 1$, as noted earlier for Gustine, Benton, and Arcata. If the annual performance database is used for calibration, a value of *P* somewhat less than 1 is found, and therefore analysis has been performed using $P = 1$. For purposes of uniformity, the presumptive C^*-values are taken to be those of Equation 8.20, leading to $C^* = 2$, 5, 10, and 20 mg/L for the four categories, respectively.

The resultant annual average *k*-values are given in Table 8.2. The median values are not much different for tertiary, secondary, and primary applications (median = 37 ± 4 m/yr), but increases for the stronger influents (super) to 189 m/yr. The spread of these distributions is quite large, implying that the characteristics of individual wetlands, or individual years in the period of record, can have strong influences on performance.

Annual Loading Relations

The BOD concentration produced in treatment wetland depends upon three primary variables (area, water flow, and inlet concentration), as well as numerous secondary variables (vegetation type, internal hydraulics, depth, event patterns, and others). It is presumed that the area effect may be combined with flow as the hydraulic loading rate (flow per unit area), because two side-by-side wetlands with double the flow should produce the same result as one at nominal flow. Therefore, two primary variables are often considered: hydraulic loading rate (q = HLR) and inlet concentration (C_i). Previous performance analyses have been based upon these two variables (Kadlec and Knight, 1996).

An equivalent approach is to rearrange the primary variables, without loss of generality, by using BLI rate ($q \cdot C_i$) and concentration (C_i). Thus it is expected that the outlet concentration produced (C_o) will depend upon BLI and C_i. A graphical display has often been adopted in the literature (Kadlec and Knight, 1996; U.S. EPA, 2000a; Wallace and Knight, 2006). In the broad context, multiple data sets are represented by a trend that shows decreasing C_o with decreasing BLI (Figure 8.9). Scatter is presumably due to secondary variable differences, such as the relative proportions of different vegetation types, hydraulic efficiencies, and other factors. The points at lowest loadings are for systems receiving very low BOD.

Each point in Figure 8.9 represents the average of one year's data for a given FWS wetland. Both BOD and CBOD data are represented; therefore, it is understood that some of the scatter is due to the difference between these two measures. The use of annual averages removes seasonal variability, if any, and precludes the effects of synoptic error (see Chapter 6).

Model Curves

The data cloud in Figure 8.9 has been reproduced in Figure 8.10, together with the *P-k-C** model results for various parameter values. The hydraulic loading is also an

TABLE 8.2
Distribution of Annual Areal Rate Coefficients k_A (m/yr) for BOD in FWS Wetlands

	Tertiary	Secondary	Primary	Super
C_i (mg/L)	0–30	30–100	100–200	>200
C^* (mg/L)	2	5	10	20
N	203	77	63	43
Percentile				
0.05	2	2	9	24
0.1	7	4	12	26
0.2	13	11	19	35
0.3	16	16	23	54
0.4	22	30	31	130
0.5	33	41	36	189
0.6	62	49	48	271
0.7	79	67	112	439
0.8	175	103	217	576
0.9	195	295	411	827

Source: The C^*-values range according to Equation 8.20, as indicated, and the value of $P = 1$.

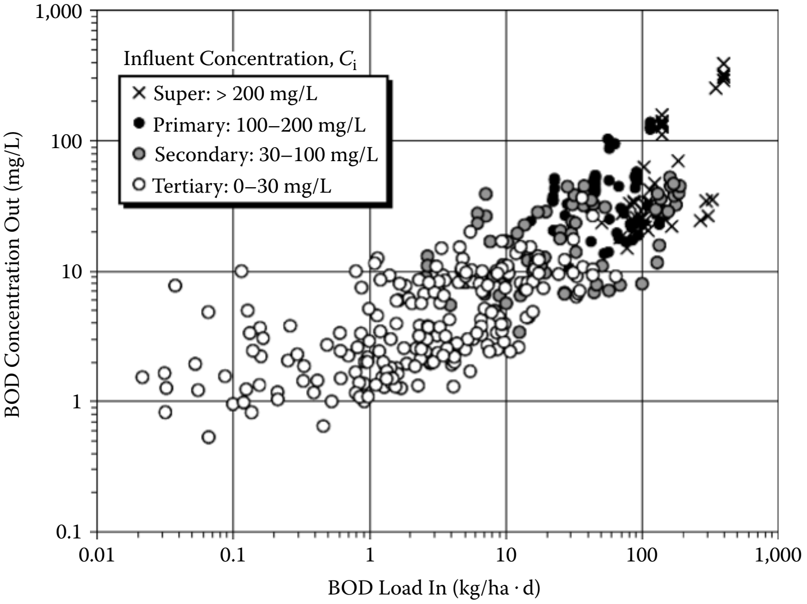


FIGURE 8.9 Outlet BOD concentration versus BOD loading for FWS wetlands. Each of the 383 points represents an annual average for one of 136 wetlands. Data groups are for tertiary ($0 < C_i < 30$ mg/L); secondary ($30 < C_i < 100$ mg/L); primary ($100 < C_i < 200$ mg/L); and "super" ($C_i > 200$ mg/L).

independent parameter in that model. It is seen that the data are bounded by Line 1, which represents high C^* and low HLR and k; and Line 2, which conversely represents low C^* and high HLR and k. These correspond to a very wide range of potential k and C^*-values; in fact, so wide that there is little resolution of the data by the model. Lines 3 and 4 represent a central tendency of the data, but do not entirely resolve either the k or C^* variability. Thus it is seen that the intersystem data

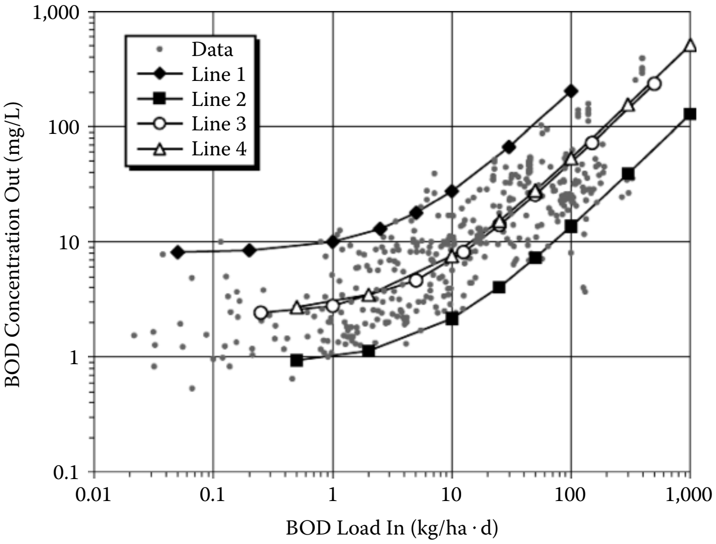


FIGURE 8.10 Selected results for the *P*-*k*-C^* model compared to annual data for BOD in FWS wetlands. The value $P = 1$ has been selected.

Line	C^* (mg/L)	k_A (m/yr)	HLR (cm/d)
1	10	15	1
2	1	250	10
3	3	60	5
4	5	35	10

does not aid in pinpointing narrow ranges of model parameters. In semiquantitative terms, the ranges that span the data are:

$$15 < k < 250 \text{ m/yr}$$

$$2 < C^* < 20 \text{ mg/L}$$

$$1 < P < 2$$

It is noteworthy that the central tendency reported by Kadlec and Knight (1996), i.e., $k = 34$ m/yr and $C^* \approx 3.5$ mg/L for $P = \infty$, is still a good central estimate for the much larger data set now available.

Variability in Annual Performances

Interestingly, the intrasystem interannual variability (year-to-year variability for one wetland with several years' data) is not necessarily much smaller than the intersystem variability (variability among several wetlands). Some single wetlands span the data cloud from one extreme to the other for different years of operation. As examples, the annual values of a few wetlands have been identified in Figure 8.11. For some, such as Poinciana, Arcata Enhancement, and Cannon Beach, the interannual variation is a significant fraction of the intersystem variation at the same loading (about 80%). Other wetlands have less interannual variability, such as Reedy Creek and Dove Creek, but still about half of the intersystem variation.

In terms of model parameters, the result is a large spread in k-values. This may be illustrated by examining the spread of k-values (for $P = 1$ and $C^* = 2$) for the various years and systems at Arcata, all working at the same site (Figure 8.12).

Out of this modeling effort, the central messages are that (1) the P-k-C^* model spans the intersystem data (as it should), but that (2) there is no resolution of the wide range of parameter values that might be selected. Consequently, the P-k-C^* model by itself is insufficient for wetland design. This simple model can be fit to a single profile or input–output data set, and represent it very well; but inherent variabilities remain quite large. It is not possible to say with certainty what next year's k-value will be, nor what the next wetland's k-value will be. Unfortunately, this is also true for C^*-values. It is informative to seek further understanding of the factors that may control performance.

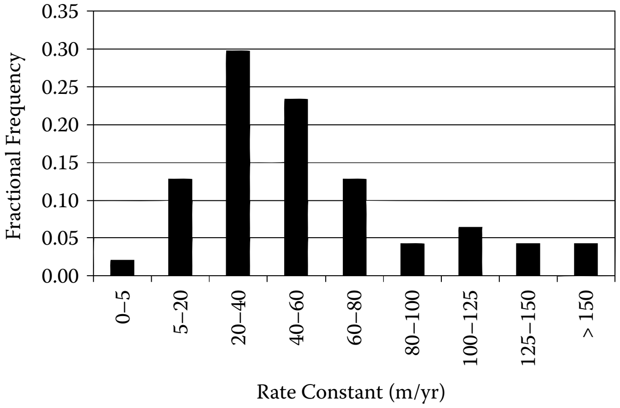


FIGURE 8.12 Rate constants for BOD removal for the aggregate of Arcata, California, data sets. The basis is $C^* = 2$ mg/L and $P = 1$. There are 23 annual average points for the pilot cells (12 cells over two years), 12 years for the combined treatment marsh cells, and 12 years for the combined enhancement marsh cells. The site $k = 54 \pm 39$ m/yr (mean ± SD).

Effects of Design and Operating Conditions

Water Depth

In Chapter 6, it was indicated that one of two assumptions were possible as limiting cases of first-order removal models:

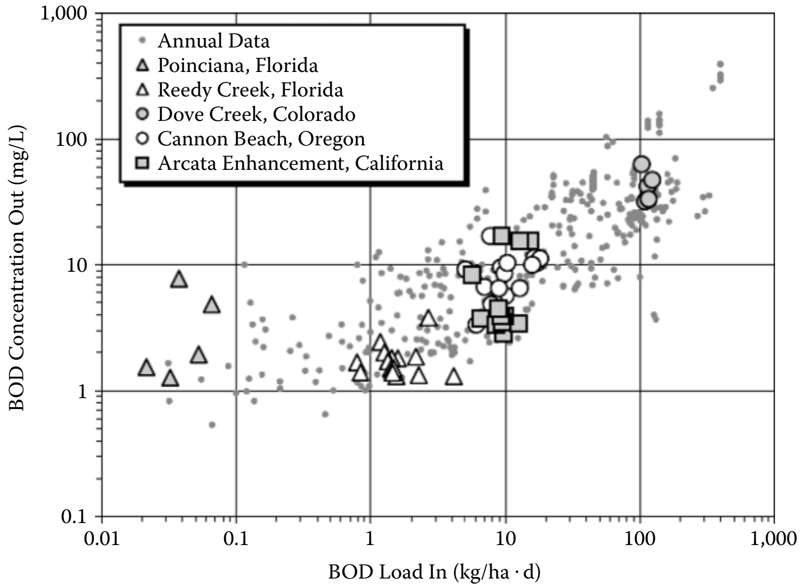


FIGURE 8.11 Single system performance within the general milieu of annual data.

either (1) the contaminant was processed everywhere within the water column, in proportion to the water volume; or (2) the contaminant was processed in proportion to the wetland planar area. In terms of model equations, the influence is exerted through the depth dependence of removal:

$$\frac{C - C^*}{C_i - C^*} = \frac{1}{(1 + k_A / Pq)^P} = \frac{1}{(1 + k_V \tau_n / P)^P} \tag{8.21}$$

from which it follows that

$$k_V = \frac{k_A}{h_n} \tag{8.22}$$

where

- h_n = nominal wetland water depth, m
- τ_n = nominal detention time, d
- k_A = areal rate coefficient, m/d (= m/yr ÷ 365)
- k_V = volumetric rate coefficient, 1/d

The question arises whether k_A is constant, or whether k_V is constant. In the former case, the extra detention time created by deeper operation is of no benefit, because k_V is reduced as depth increases; in the latter case, increased depth creates no penalty in decreased k_V-values, and performance can be increased by increasing the water depth.

As one test of the two possibilities, operational data from a wetland with sequentially varied depths may be examined. The Listowel wetlands were operated at various depths over a four-year period, with the resulting ability to examine Equation 8.22. There is a strong increase in k_V-values with ($1/h_n$) for depths above about 5 cm (Figure 8.13), indicating that k_A is more nearly constant than k_V. It is possible that the drop in k_V for depths less than 5 cm is due to the incomplete wetting of the wetland surface.

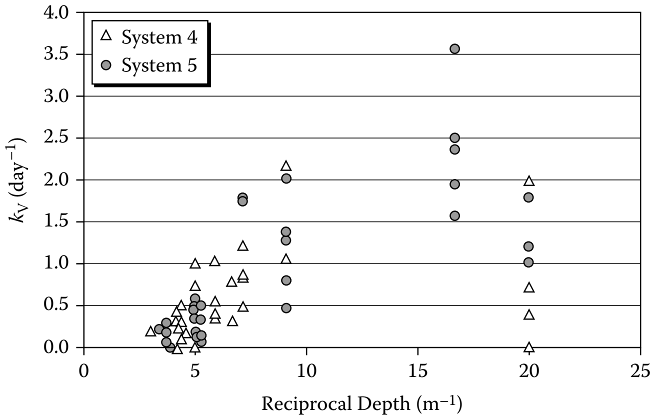


FIGURE 8.13 Variation of the volumetric rate constant for BOD removal for Listowel, Ontario, Systems 4 and 5. The parameters P = 2 and C^* = 2 mg/L have been chosen.

A second test is to compare side-by-side wetlands operated at different depths. The Arcata pilot wetlands were operated in that fashion for two years. Each of three hydraulic loadings was replicated at two depths. For each loading, the value of k_V was lower at the larger depth (Table 8.3). Over the entire suite of experiments, a 35% depth increase resulted in a 35% k_V decrease. This also indicates that k_A is more nearly constant than k_V.

Either k_A or k_V can be used to represent a data set or be used in design. However, the use of k_V requires the accompanying information on water depth (h) because of the depth dependence indicated in Equation 8.22. This depth dependence also means that more detention time created by deeper water is counteracted by a decrease in the volumetric rate constant. The hydraulic loading rate is not depth-dependent,

TABLE 8.3
Depth Effects on Rate Constants for the Arcata, California, Pilot Project

HLR (cm/d)	Depth (cm)	Percent Depth Increase	k_V (1/day)	Percent k_V Decrease
0.230 ± 0.010	36 ± 2.0	—	0.71	—
0.215 ± 0.025	52 ± 0.5	31%	0.58	18%
0.110 ± 0.005	27 ± 0.5	—	0.52	—
0.113 ± 0.003	46 ± 2.0	41%	0.26	50%
0.065 ± 0.005	30 ± 2.0	—	0.39	—
0.065 ± 0.005	46 ± 0.5	35%	0.25	36%
Mean		**36%**		**35%**

Note: Twelve pilot cells were operated as duplicates at two depths and three hydraulic loading rates, over a period of two years, beginning one year after start-up. The P-k-C^* model parameters were fixed at P = 1 and C^* = 2 mg/L.

Source: From analysis of data in Gearheart *et al.* (1989). In *Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural*. Hammer (Ed.), Lewis Publishers, Chelsea, Michigan, pp. 121–137.

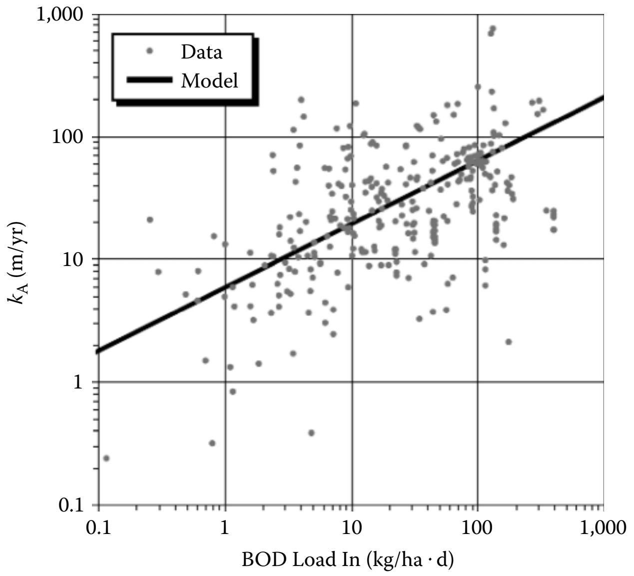


FIGURE 8.14 Dependence of rate constants on BOD loading.

and the same data indicate that k_A is nearly independent of depth. The use of areal coefficients does not require depth. For many FWS wetlands, especially large ones, depth is not known to a reasonable degree of accuracy (see Chapter 2). For these reasons, the parameter k is used herein.

Loading Effect on *k*-Values

Importantly, both k_V and k depend to some degree upon BLI rate. This is the observed trend of the data from a large number of free water surface wetlands (Figure 8.14). The selected parameters were $P = 2$ and $C^* = 2$ mg/L. Although the correlation depends to some extent upon the values of P and C^*, there is no selection of these parameters that removes the dependence of k_V and k on the BLI rate.

The first-order model has increased sensitivity to loading if the value of C^* is chosen to be zero (Kadlec, 2000). Under that assumption, the values of k_{V1} are nearly proportional to BLI, or inversely proportional to the detention time, for low hydraulic loadings. The additional subscript "1" indicates that the model contains only one parameter, the k-value, as opposed to two (k and C^*). This sensitivity is exacerbated if the plug flow model is used, i.e., $P = \infty$. The near-proportionality of k_{V1PF} to BLI has been repeatedly recognized (Reed *et al.*, 1995; Kadlec, 2000; Water Environment Federation, 2001; Ran *et al.*, 2004). WEF (2001) report the following relation:

$$k_{V1PF} = 0.030 + 0.00648 \cdot \text{BLI} \tag{8.23}$$

where

BLI = BOD loading in, kg/ha·d

k_{V1PF} = plug flow rate constant with $C^* = 0$, d^{-1}

This dependence leads to a design paradox. The required wetland area is inversely proportional to the k-value, whereas the inlet BLI is inversely proportional to wetland area. Suppose a BLI has been chosen as a first estimate, and the corresponding k-value determined (e.g., from Equation 8.23); and the predicted outlet BOD is too high. The obvious correction is to increase area. However, that lowers the inlet BLI, and according to Equation 8.23, also lowers the k-value. Clearly, this is a useless procedure. Reed *et al.* (1995) dispose of the difficulty by ignoring Equation 8.23. This regression is an example of the spurious correlation caused by hydraulic loading appearing in both the abscissa and ordinate (see Chapter 6).

Temperature

The first-order model has been reliable for predicting removal rates of organic matter in most wastewater treatment processes (Metcalf and Eddy Inc., 1991). The modified Arrhenius relationship is commonly used to adjust the removal rate coefficient for temperature in traditional wastewater treatment processes:

$$k_{V1} = k_{V1,20}\theta^{(T-20)} \tag{8.24}$$

where

k_{V1} = rate constant at temperature T, d^{-1}

$k_{V1,20}$ = rate constant at 20°C, d^{-1}

T = water temperature,°C

θ = modified Arrhenius temperature factor, dimensionless

Values of θ for various treatment technologies range from 1.00 to 1.08, with typical values of 1.04 for activated sludge, 1.08 for aerated lagoons, and 1.035 for trickling filters (Metcalf and Eddy, Inc., 1991). The temperature dependence of the BOD test itself is generally taken to be 1.047 (Crites and Tchobanoglous, 1998). These traditional process units differ considerably from wetlands, in terms of functional complexity and operating conditions. They are designed to provide intense focus on microbial processes alone, without other biotic components or the spatial heterogeneity of a treatment wetland.

The treatment wetland literature is replete with the assertion that a θ-value of about 1.06 applies to FWS wetlands (Reed *et al.*, 1988: 1.10; U.S. EPA, 1988b: 1.10; Reed *et al.*, 1995: 1.06; Crites and Tchobanoglous, 1998: 1.06; Campbell and Ogden, 1999: 1.06; U.S. EPA, 2000a: 1.04). These reports all referred to the plug flow model with $C^* = 0$. However, Kadlec and Knight (1996) could not find a temperature dependence in wetland BOD data. That finding was subsequently supported by analysis of more systems (Kadlec and Reddy, 2001).

The two most closely related companion technologies for BOD reduction are overland flow and stabilization ponds. The former involves very shallow (a few centimeters depth at most) water flow over a vegetated surface, and the latter represent algal-aquatic systems with typical depths of one to two meters. Thus, these technologies may be regarded as the shallow- and deepwater extremes of treatment wetlands. The data from those systems yield temperature coefficients that are close to 1.00 for ponds (1.005 ± 0.014) and overland flow (1.01 ± 0.01) (Kadlec and Reddy, 2001). U.S. EPA (1983a) suggests several different design approaches for facultative ponds, including equivalents to the first-order model presented above. The suggested design temperature factors are θ = 1.085 and 1.090. However, U.S. EPA (1983a) show a data basis that produces θ = 0.995. The authors explain this as follows: "The logical explanation for the lack of influence by temperature is that the pond systems are so large that the temperature effect is masked by other factors." No explanation was offered for rejecting the observed behavior in the recommended design calculations. This lack of a temperature effect in ponds has more recently been reported by Abis (2002).

Here the temperature effect on performance of several wetland systems has been re-analyzed with the *P-k-C** model, with $P = 1$ (Table 8.4). The θ-value is 0.985 ± 0.021, meaning slightly worse performance at higher temperatures. Little or no variance is removed by adding a θ-factor to the model. It is clear that the complex of wetland ecosystem processes is masking the known microbial temperature sensitivity expected for suspended growth systems. One candidate explanation is oxygen transfer, which must be adequate to justify the first-order approximation. However, as seen in the earlier section on carbon processing, many other processes can influence BOD removal.

The preponderance of evidence suggests that wetland BOD removal is not improved at higher wetland water temperatures.

Seasonal Trends

There are typically gentle annual cycles in the effluent BOD from FWS wetlands (Figure 8.15). A maximum is seen in spring or summer, and the amplitude of the annual cycle is on the order of 30% of the mean (Table 8.5). The trend is described by

$$C = C_{avg}\left[1 + A\cdot\cos\left(\omega(t - t_{max})\right)\right] \tag{8.25}$$

where

A = trend fractional amplitude, dimensionless
C = concentration, mg/L
C_{avg} = mean annual concentration, mg/L
t = yearday, d
t_{max} = yearday for maximum concentration, d
ω = annual period, 0.01721 d^{-1}

These cycles often do not reflect contemporary influent BOD or the contemporary hydraulic loading to the wetland. This is evidenced in Figure 8.15, where minima of the inlet concentration correspond to maxima of the outlet concentration, for relatively uniform hydraulic loading throughout the year.

TABLE 8.4
Temperature Factors for the *P-k-C Model for Example FWS Wetlands**

Wetland	Cell	Data (years)	k_{20} (m/yr)	C^* (mg/L)	θ
Brighton, Ontario	1	4	25	4	0.946
Columbia, Missouri	1	2	450	8	0.996
Listowel, Ontario	1	4	30	2	1.002
	2	4	36	5	1.035
	3	4	19	3	0.932
	4	4	89	5	0.986
	5	4	49	6	0.977
Arcata, California	1	2	51	0	0.993
	2	2	92	9	0.973
	3	2	44	4	0.993
	4	2	56	4	0.999
	5	2	60	6	0.978
	6	2	76	3	0.988
	7	2	33	0	0.989
	8	2	50	11	0.999
	9	2	25	0	0.980
	10	2	23	0	0.975
	11	2	54	4	0.992
	12	2	73	5	0.976
Mean					**0.985**

Note: The value $P = 1.0$ has been selected. Model fits are not good, in the sense that R^2-values do not increase much when a θ-factor is added.

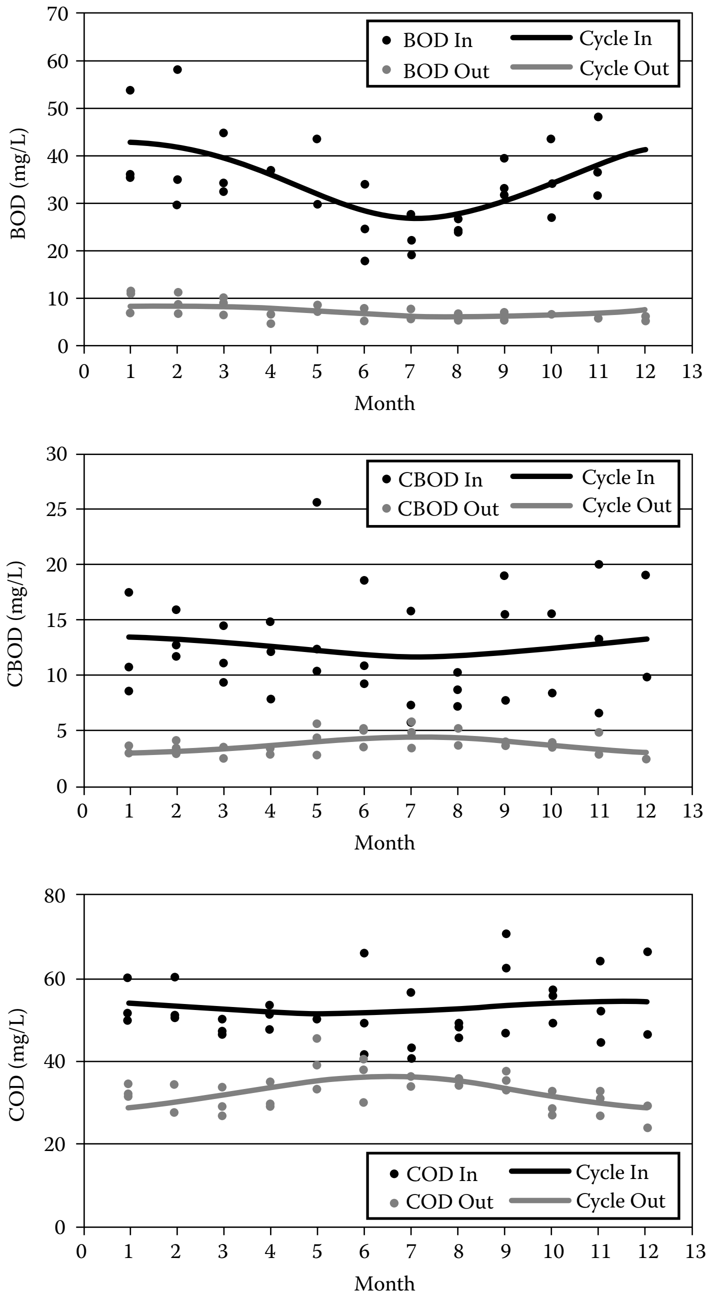


FIGURE 8.15 Annual cycles of BOD, CBOD, and COD for three years of monthly averages of daily data from Columbia, Missouri.

The considerable scatter in effluent concentrations contributes to low R^2-values for the trend lines (Table 8.5). This behavior is of concern in wetland sizing, if the peak values of the concentrations are of importance in the regulatory compliance for the project.

Variability around Seasonal Trends

Because stochastic behavior is present in moderate amount, it is necessary to quantify performance variability, and ultimately to modify sizing based upon that understanding. Average effluent BOD values over short time periods are subject to variation from the annual mean. The longer the averaging period, the closer the short-term mean value is to the annual mean value. For FWS wetlands, average effluent BOD concentrations are distributed approximately according to the log normal distribution. Examples of these distributions are given in Figure 8.16.

The averaging period has a very strong influence on the higher percentiles, which form the basis for permit requirements. The example given in Figure 8.17 shows that for the Columbia, Missouri, system, the daily maximum is about triple the monthly maximum, and the weekly maximum is about double the monthly maximum. These ratios shrink as

TABLE 8.5
Sinusoidal Trends in FWS Wetland Effluent BOD Concentrations during the Course of the Year

Site	POR* Months	Operation Period	Sample Frequency	Averaging Period	Trend Mean (mg/L)	Trend Fractional Amplitude	Trend t_{max} (Julian day)	Trend R^2
BOD								
Estevan	59	Summer	Weekly	None	4.8	0.31	201	0.02
Listowel 1	48	Annual	Weekly	Monthly	8.1	0.36	78	0.12
Listowel 2	48	Annual	Weekly	Monthly	11.0	0.55	67	0.22
Listowel 3	48	Annual	Weekly	Monthly	7.3	0.44	67	0.40
Listowel 4	48	Annual	Weekly	Monthly	9.5	0.38	145	0.13
Listowel 5	48	Annual	Weekly	Monthly	14.1	0.30	67	0.10
Cannon Beach	192	Dry season	Monthly	Monthly	7.3	0.12	105	0.11
Columbia	36	Annual	Daily	Monthly	7.3	0.16	54	0.27
CBOD								
Columbia	15	Annual	Daily	None	10.8	0.31	114	0.25
Columbia	15	Annual	Daily	Weekly	10.9	0.32	122	0.41
Columbia	15	Annual	Daily	Monthly	10.8	0.31	116	0.71
Brighton	39	Annual	Weekly	Monthly	4.4	0.29	18	0.40
Orlando Easterly	120	Annual	3× Monthly	3× Monthly	0.9	0.09	62	0.01
Tres Rios H1	84	Annual	Weekly	Monthly	2.9	0.53	215	0.12
Tres Rios H2	84	Annual	Weekly	Monthly	2.4	0.23	190	0.07
Arcata Treatment	156	Annual	Weekly	Weekly	23.2	0.13	285	0.04
Arcata Enhancement	120	Annual	Weekly	Weekly	3.8	0.33	22	0.12
TOC								
Estevan	59	Summer	Weekly	None	18.4	0.21	251	0.06
Orlando Easterly	120	Annual	3× Monthly	3× Monthly	10.2	0.11	160	0.22
Tres Rios H1	84	Annual	Weekly	Monthly	8.5	0.17	164	0.20
Tres Rios H2	84	Annual	Weekly	Monthly	7.9	0.13	124	0.34
COD								
Tres Rios H1	84	Annual	Weekly	Monthly	42.2	0.38	156	0.28
Tres Rios H2	84	Annual	Weekly	Monthly	41.7	0.23	132	0.06
Columbia	36	Annual	Daily	Monthly	32.6	0.12	179	0.38

* POR = period of record

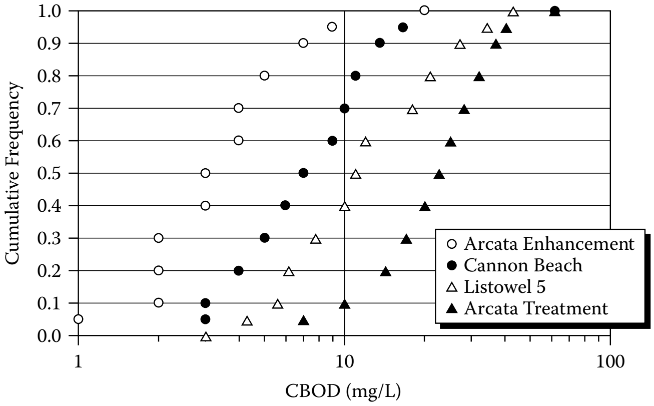


FIGURE 8.16 Frequency distributions for monthly BOD (Cannon Beach, Oregon; Listowel, Ontario) and weekly CBOD (Arcata, California).

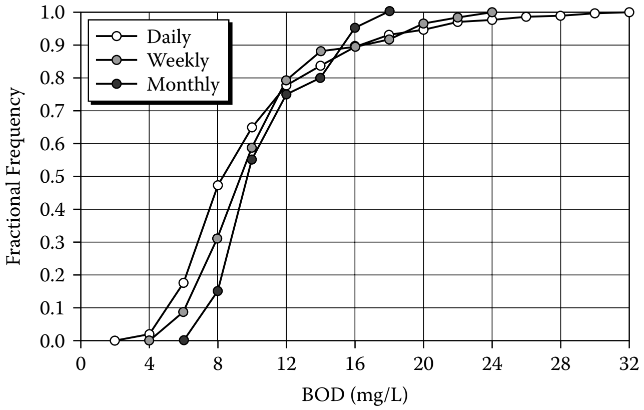


FIGURE 8.17 Frequency distributions at the Columbia, Missouri, wetlands for daily (five days out of seven), weekly (average of five dailies), and monthly BOD values (average of 22 dailies). The 90th percentile is about 1.6 times the mean. However, the maximum daily value is about triple the mean.

TABLE 8.6
Trend Multipliers for Effluent BOD Distributions in FWS Wetlands

Site	Trend Mean (mg/L)	Trend Multiplier (80th percentile)	Trend Multiplier (90th percentile)	Trend Multiplier (95th percentile)	Trend Multiplier (100th percentile, maximum)
BOD					
Estevan, Saskatchewan	4.8	1.53	2.37	2.88	3.68
Listowel 1, Ontario	8.1	1.66	1.85	2.36	2.74
Listowel 2, Ontario	11.0	1.46	1.82	2.20	3.26
Listowel 3, Ontario	7.3	1.60	2.26	2.78	3.45
Listowel 4, Ontario	9.5	1.46	1.97	2.51	2.90
Listowel 5, Ontario	14.1	1.51	2.15	2.35	2.60
Cannon Beach, Oregon	7.3	1.39	1.60	1.88	2.39
CBOD					
Columbia, Missouri	10.8	1.25	1.55	1.72	2.41
Columbia, Missouri	10.9	1.25	1.38	1.51	1.65
Columbia, Missouri	10.8	1.15	1.18	1.19	1.20
Brighton, Ontario	4.4	1.32	1.55	1.75	2.09
Orlando Easterly, Florida	0.9	1.30	1.45	1.57	1.80
Tres Rios H1, Arizona	2.9	1.43	1.96	2.37	4.11
Tres Rios H2, Arizona	2.4	1.44	1.77	2.18	3.49
Arcata, California Treatment	23.2	1.24	1.35	1.42	1.66
Arcata, California Enhancement	3.8	1.36	1.80	2.24	4.98
TOC					
Estevan, Saskatchewan	18.4	1.30	1.37	1.56	2.80
Orlando Easterly, Florida	10.2	1.09	1.15	1.21	1.36
Tres Rios H1, Arizona	8.5	1.10	1.27	1.36	1.74
Tres Rios H2, Arizona	7.9	1.11	1.15	1.18	1.42
COD					
Tres Rios H1, Arizona	42.2	1.34	1.56	1.72	2.10
Tres Rios H2, Arizona	41.7	1.16	1.47	1.80	4.52
Average All		**1.33**	**1.56**	**1.78**	**2.51**

Note: The corresponding trend line information is given in Table 8.5. Trend multiplier is $(1+\Psi)$; see Equation 6.61.

the percentile is relaxed to 90th, where they are all about equal.

Example seasonal trend information has been given (Table 8.5). An exploration of the scatter of data around those trends establishes the percentile rank of multipliers to the trend values. Table 8.6 shows that the carbon compound multiplier of the trend is 1.78. That means that excursions in individual monthly samples greater than 78% higher than the trend may be expected in 1 month out of 20.

Model Dynamics

The response of a FWS wetland to changes in operating conditions does not necessarily follow a first-order model, or any other deterministic model that pertains only to the surface water body. Changes in inlet concentrations, for instance, may not be reflected in outlet concentrations, even if allowance is made for transport delay. For example, the Columbia wetlands had a detention time of three to four days, and experienced month-to-month variations in inlet BOD spanning 10–50 mg/L (Figure 8.18). During the same period, outlet BOD ranged from 5 to 30 mg/L, but there is not a correspondence between peaks, i.e., there is no tracking of the inlet to be seen in the outlet.

The conclusion is that no deterministic removal model now available in the literature (*P-k-C** model included) should be used to predict high frequency BOD events, even to the scale of monthly variations.

Oxygen Supply

If removal of BOD is via Equation 8.10, oxygen transfer must be adequate to justify the first-order removal approximation. However, processes detailed in Equations 8.11 through 8.18 can also influence BOD removal, especially in heavily loaded systems. Fermentation, nitrate, iron, and sulfate reduction are all potential consumers of carbon compounds in the absence of free oxygen. Ultimately, under very low redox conditions, methanogenesis may take place. Therefore, the implied

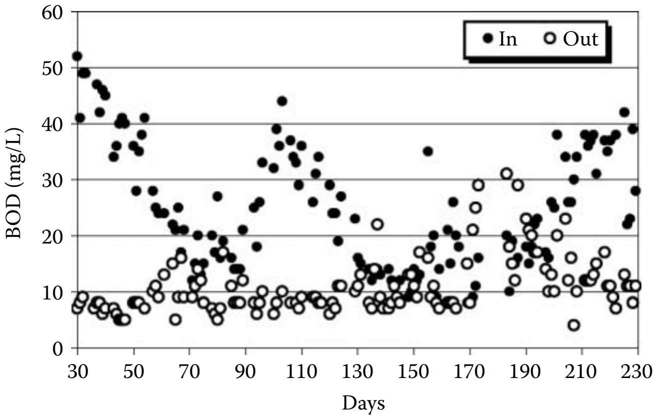


FIGURE 8.18 Daily BOD for a 200-day period at Columbia, Missouri, commencing October 24, 1994. The hydraulic loading rate during this period was relatively steady at 15 ± 2 cm/d, equivalent to a nominal detention time of about three days.

maximum oxygen supply for BOD removal is simply the load of BOD removed. The systems that form the basis for Figure 8.9 have the median oxygen requirements shown earlier in Table 8.7. The supply to the water in FWS wetlands is likely to be no more than 2–4 g/m²·d (see Chapter 5). Therefore, as the incoming BOD increases to the levels seen in primary and super treatment situations, it is unlikely that oxidative processes are entirely responsible for the destruction of BOD compounds. Additional mechanisms, such as anaerobic digestion (methanogenesis) become important contributors to removal. It is tempting to speculate that aeration of the wetland water may be enhanced by open water sections, but that is a questionable hypothesis as seen in the next section.

Open Water Fraction

BOD is reduced in both ponds and wetlands. However, there are differences in several aspects of these systems that argue for differences in their relative BOD removal capabilities. The loading graph may be used to explore intersystem effects of open water. In a broad context, multiple data sets are represented by trends that show decreasing C_o with decreasing BLI (see Figure 8.19). For BLI less than about 100 kg/ha·d, there appears to be little difference between ponds and wetlands for BOD removal (Figure 8.19). At higher loadings, there is a strong suggestion that ponds are better than wetlands, although wetland data is sparse at high loadings (Kadlec, 2005e). It is perhaps ironic that the upper BLI limit sometimes imposed for pond operation of 80–90 kg/ha·d (Shilton, 2005; Crites *et al*., 2006) represents the lower limit for which pond performance is distinctly better than wetland performance.

Open water areas have been suggested as necessary and optimal for BOD reduction in FWS systems, for loadings up to 60 kg/ha·d (U.S. EPA, 2000a). Performance data do not support that hypothesis (Figure 8.20). However, open water zones do not appear to impair BOD removal.

8.3 BOD REMOVAL IN HSSF WETLANDS

A large amount of BOD data now exists for HSSF wetlands, mostly treating domestic wastewaters. The same ways are used to summarize this information as for FWS wetlands, including removal rate models and graphical summaries. As for FWS systems, when waters with moderate to large concentrations of BOD flow through a HSSF wetland, a decrease in concentration to a nonzero plateau is typically observed. This behavior is illustrated in Figure 8.21 for two continuous flow

TABLE 8.7
Load Reduction of BOD_5 in FWS Wetlands

	Tertiary (g/m²·d)	Secondary (g/m²·d)	Primary (g/m²·d)	Super (g/m²·d)
C_i =	3–30 mg/L	30–100 mg/L	100–200 mg/L	>200 mg/L
N =	204	77	63	43
Percentile				
0.05	0.11	0.19	1.32	6.37
0.10	0.15	0.30	1.52	7.24
0.20	0.27	0.56	2.18	7.75
0.30	0.45	0.71	2.49	8.36
0.40	0.51	0.85	2.63	8.89
0.50	0.66	1.33	2.88	9.39
0.60	0.80	1.55	3.97	9.74
0.70	0.93	2.35	4.99	10.84
0.80	1.08	3.74	5.74	23.83
0.90	1.45	5.11	8.10	26.42
0.95	2.07	8.85	10.15	27.33

Note: These amounts are the implied oxygen requirement for aerobic destruction of the compounds that comprise BOD_5. N represents wetland·years.

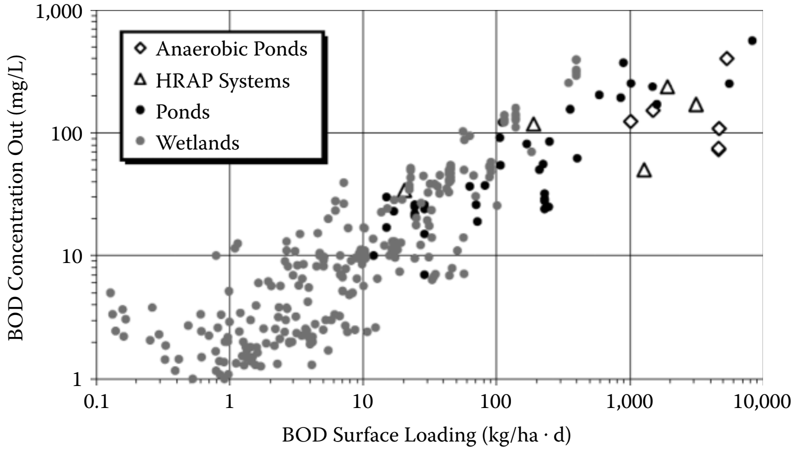


FIGURE 8.19 Response annual average effluent BOD of aquatic systems to increasing annual average BOD loadings. Wetlands are represented by 265 years of data for 113 systems. Pond data are for 51 systems over their period of data record. Wetland data are from the North American Database (1998); together with unpublished data. Pond data are from U.S. EPA (1983a), Mendes *et al*. (1994), Pearson *et al.* (1995), Soler *et al*. (1995), El Hamouri *et al.* (1995), Abis (2002), Tadesse *et al.* (2003), Craggs *et al.* (2003); together with unpublished data.

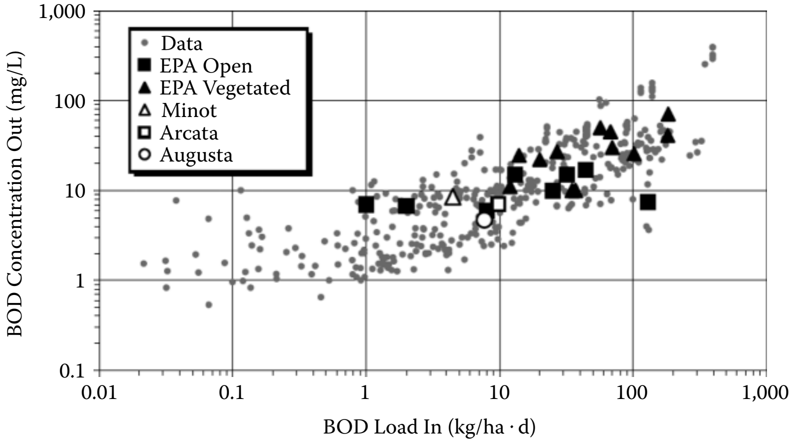


FIGURE 8.20 Wetlands with open water sections. The solid points are plotted from U.S. EPA (2000a). The open points represent wetlands built with large open water components in their central region. Dots are the general milieu of FWS performances.

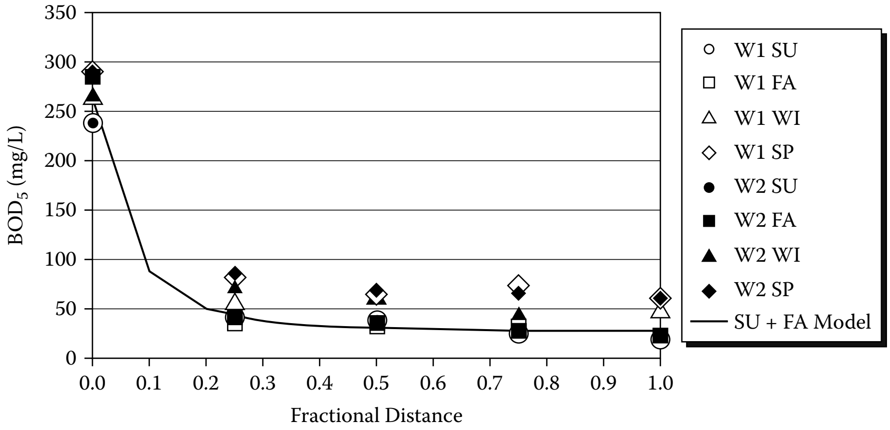


FIGURE 8.21 Longitudinal profiles of BOD_5 at the two NERCC, Minnesota, HSSF wetlands (W1 & W2), over a two-year period of record, by quarter. Note the plateau concentration is somewhat higher in winter and spring. The *P-k-C** model is shown as the solid line, for the summer and fall period. The fit values are $P = 4$, $k = 66$ m/yr, $C^* = 27$ mg/L, with an $R^2 = 0.998$.

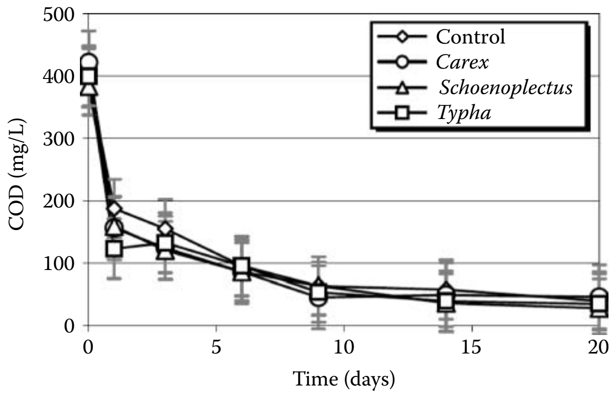


FIGURE 8.22 Reduction of COD in planted and unplanted batch SSF gravel mesocosms. The pollutant solution was concocted from meat protein and sucrose. (From Stein *et al.* (2006a) *Ecological Engineering* 26(2): 100–112. Reprinted with permission.)

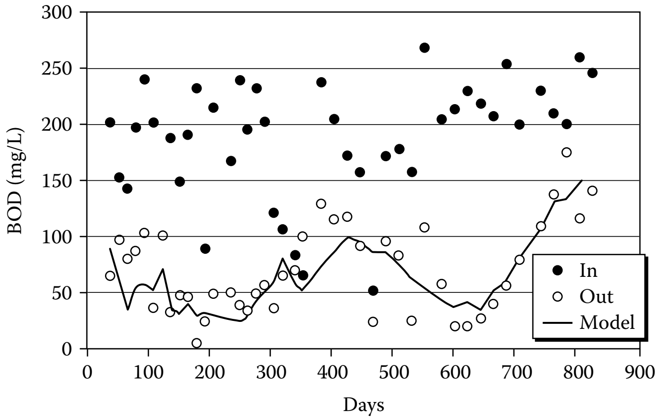


FIGURE 8.23 First-order *P*-*k*-*C** fit for Grand Lake, Minnesota. The value of $P = 2$, $k_{20} = 33$ m/yr. The fitting process is insensitive to $C^* = 20$ mg/L. The temperature coefficient is $\theta = 1.140$, the highest of any system studied here. The value of $R^2 = 0.93$. (From unpublished data.)

HSSF wetlands near Duluth, Minnesota (NERCC project, described in Kadlec *et al.*, 2003). Samples were taken along the wetland length, from internal wells, and the midpoint transfer between the two cells in series for each of the two systems. The same sort of response is seen in batch loaded systems (Stein *et al.*, 2006a, b). Typical response data showed a sharp decrease in BOD_5 and COD to a nonzero background (Figure 8.22). The decrease in these batch experiments is reasonably well fit by the *k*-*C** model.

In more general terms, the *P*-*k*-*C** first-order model can readily account for these observations, for appropriate values of parameters (see Chapter 6). However, as for FWS wetlands, parameter values are known to depend on system hydraulics (Kadlec, 2000), as well as on speciation of the BOD (Crites and Tchobanoglous, 1998; Kadlec, 2003a).

First-Order Modeling

The considerations of weathering as well as speciation of BOD and chemical oxygen demand (COD) that were discussed for FWS wetlands also apply for HSSF systems. It is anticipated that the *P*-*k*-*C** model (Equation 8.20) will apply, with the parameter *P* being somewhat less than the *N*TIS value determined for a nonreactive tracer (Kadlec, 2003a).

The parameter *P* accounts for two effects: the detention time distribution (DTD) and the *k*-value distribution (*k*VD) (see Chapter 6). The value of *P* is always less than the number of tanks determined from a tracer test. For broad distributions of *k*-values, such as may occur for BOD, the typical HSSF hydraulic TIS number of six to ten (see Table 6.2) will be reduced to a *P*-value of three or four. The *C**-value in Equation 8.20 reflects several possible different causes for HSSF, as for FWS. A number of different approaches to data fitting may be used.

Reasonable data fits may be obtained for time series for specific wetlands. As an example, the data from Grand Lake, Minnesota, are shown, along with the model fit, for a two-year period of record (see Figure 8.23). This wetland was tracer-tested, and produced *N*TIS = 3.3. This value was reduced to $P = 2$ for the fitting process. The major time trend is captured, but considerable scatter remains. In order that both *k* and *C** can be determined with a good degree of certainty, the wetland must experience significant changes in loadings and concentrations over the course of time. If the wetland is operated in the batch mode, it is reasonable to expect that the exponential ($P = \infty$) form of Equation 8.20 should be used, perhaps after decrementing *P* for a possible *k*VD (Stein *et al.*, 2006a). An example of an exponential fit is shown in Figure 8.23.

Interior distance profiles may be fit for a given wetland. Figure 8.21 displays such a fit for the NERCC wetlands near Duluth, Minnesota. This system had two cells in series, in each of two trains. Therefore, the value $P = 4$ was assigned. The values $k = 66$ m/yr and $C^* = 27$ mg/L fit the data quite well ($R^2 = 1.00$), but the interior points are sparse. The profile shows that a plateau is reached in the front end of the train, with most of the system exhibiting a nondecreasing concentration. This is an extremely important feature of HSSF systems, because it suggests that the use of output information will often reflect the background concentration, and not contain any information on the drop-off to that outlet (plateau) concentration. One disadvantage of the profile fitting method is that it requires extensive interior monitoring, which is often not feasible. A second disadvantage is that interior sample points may not be situated in an "average" part of the flow path. If there are cells in series (there were two at NERCC), the transfer structure will provide a flow-weighted sample, but interior sample points do not necessarily do the same. For instance, attempts to sample at three cross-cell positions, and three distances, at the Benton, Kentucky, facility produced no consistent patterns (TVA, 1990). Multiple internal sample points at Minoa, New York, in three dimensions, also produced

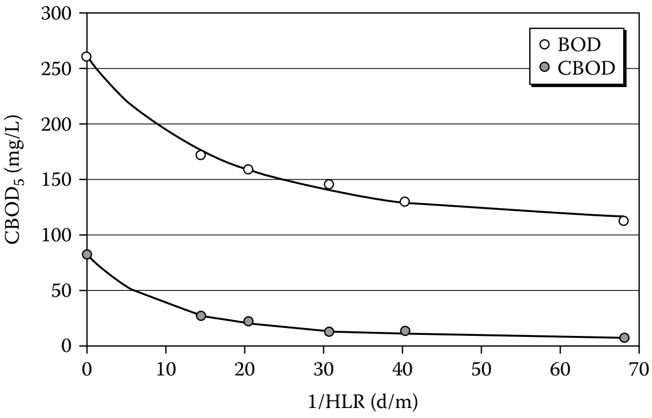


FIGURE 8.24 Reduction of $CBOD_5$ and BOD_5 in side-by-side *Schoenoplectus* SSF gravel wetlands operated at different hydraulic loading rates. The pollutant solution was concocted from meat protein and sucrose. The model lines are for $P = 4$, $C^* = 6.6$ mg/L, and $k = 38$ m/yr ($R^2 = 0.94$) for $CBOD_5$, and $C^* = 106$ mg/L and $k = 22$ m/yr ($R^2 = 0.97$) for BOD_5. (Data from Tanner *et al.* (1998b) *Journal of Environmental Quality* 27(2): 448–458.)

TABLE 8.8
First-Order Areal *k*-Values for HSSF Wetlands, Based Upon Period of Record Input–Output Analysis

	Tertiary	Secondary	Primary	Super
C_i =	3–30	30–100	100–200	>200
C^* =	1	5	10	15
P =	3	3	3	3
N =	52	53	51	27
Percentile				
0.05	11	5	9	3
0.1	15	16	10	9
0.2	25	20	12	14
0.3	36	24	15	21
0.4	63	30	23	33
0.5	86	37	25	66
0.6	154	39	28	98
0.7	224	44	44	114
0.8	287	82	62	210
0.9	458	167	107	378
0.95	703	228	132	447

Note: The number of wetlands in each category is N.

erratically variable results from which model parameters cannot reliably be determined (Theis and Young, 2000).

Side-by-side wetlands may be operated at different hydraulic loading rates. These will experience the same inlet concentrations and meteorology, but will be subject to slight unavoidable differences in ecology. Figure 8.24 shows such a fit for the Hamilton, New Zealand, wetlands, which received dairy parlor effluent (Tanner *et al.*, 1998b). This system had five cells in parallel, planted with *Schoenoplectus* (*Scirpus*) *tabernaemontani*. As the hydraulic loading rate is decreased (nominal HRT increases), the outlet concentration for the two-year period of record decreases to a plateau value, identified with C^*. This technique is obviously a research tool only, because multiple wetlands at different loading rates will not be built for routine service.

Long-term average input–output data may be used to estimate *k*-values for assumed values of P and C^*. However, if the outlet concentration is close to or at C^*, the estimate will be a lower bound. Under that circumstance, the concentration may have dropped to near C^* well before the system effluent point, meaning the *k*-value could have been much higher. As an example, see Figure 8.21. There are input–output data at the end of Cell 1 ($x = 0.5$), and at the end of Cell 2 ($x = 1.0$). Analysis, using $C^* = 27$ mg/L, shows:

$$\text{Profile fit: } k = 66 \text{ m/yr}$$

$$\text{Cell 1 I/O fit: } k = 23.5 \text{ m/yr}$$

$$\text{Cells 1+2 I/O fit: } k = 16.4 \text{ m/yr}$$

This pitfall can be avoided only if the outlet concentration is well above the presumptive C^*-value. The reader is referred to Kadlec (2000) for further details of such potential misleading interpretations, and to the discussion in Kadlec and Knight (1996). To summarize, the first-order model appears to be perfectly capable of describing BOD profiles and time series in HSSF wetlands. However, if only input–output data are analyzed, there is a strong chance that *k*-values will be lower than those from longitudinal transects. In turn, it implies that extrapolation to lower loading rates will be risky, although extrapolation to higher loading rates will be overly conservative. As an indicator of the *k*-values to be expected, Table 8.8 shows the percentile points of distributions of long-term average input–output *k*-values for HSSF for selected C^* and $P = 3$.

It is noteworthy that the central tendency reported by Kadlec and Knight (1996), i.e., $k = 180$ m/yr and $C^* \approx 3.5$ mg/L for $P = \infty$, is not a good estimate for the much larger data set now available. Depending on the strength of the wastewater being treated, *k*-values are lower, and have a broad intersystem distribution of values.

Loading Effect on *k*-Values

As a consequence of the plateau effect, the *k*-values for BOD are hydraulic load-dependent (Figure 8.25). The values of k are nearly proportional to hydraulic loading or inversely proportional to the detention time. The same result holds for dependence on the BLI to the wetland. The near-proportionality of k_{V1PF} to BLI has been repeatedly recognized (Reed *et al.*, 1995; Kadlec, 2000; Water Environment Federation, 2001; Ran *et al.*, 2004). WEF (2001) report the following relation for HSSF wetlands:

$$k_{V1PF} = 0.050 + 0.01054 \cdot \text{BLI} \tag{8.26}$$

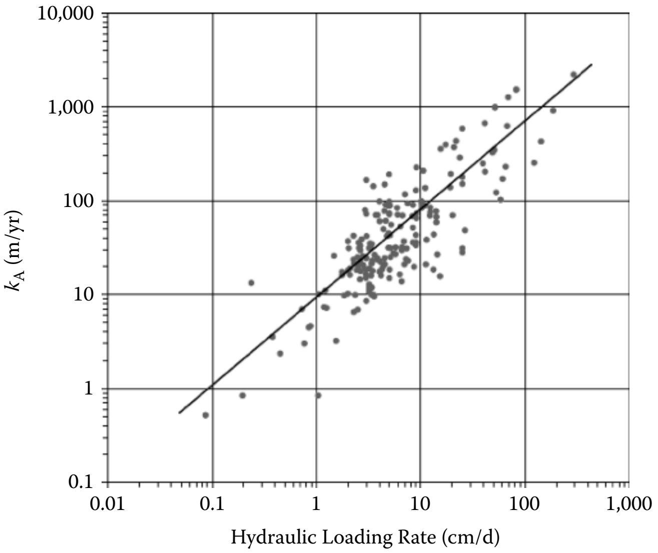


FIGURE 8.25 Dependence of the first-order areal rate constant on hydraulic loading. The values $P = 3$ and $C^* = 2$ mg/L have been used. The trend line has $R^2 = 0.76$.

where

BLI = BOD Loading In, kg/ha·d

k_{V1PF} = plug flow rate constant with $C^* = 0$, d^{-1}

As noted in the section on FWS in this chapter, this dependence leads to a design paradox. This graph is also subject to the spurious effect of containing the hydraulic loading in both the abscissa and ordinate.

Graphical Relations

The graphical display that has often been adopted in the literature (Kadlec and Knight, 1996; U.S. EPA, 2000a; Wallace and Knight, 2006) plots outlet BOD concentrations versus inlet BLI (Figure 8.26). In the broad context, the multiple data sets are represented by a trend that shows decreasing outlet concentration with decreasing BLI, but that relationship is obscured by large scatter. Each point in Figure 8.26 represents the average of the entire period of record data for a given HSSF wetland. Both BOD and CBOD data are represented; therefore, it is understood that some of the scatter is due to the difference between these two measures. The use of period of record averages removes seasonal variability, if any, and precludes the effects of synoptic error (see Chapter 6).

A second display is outlet concentration versus inlet concentration (Figure 8.27). This graph shows a more consistent central trend, with a log-linear regression coefficient

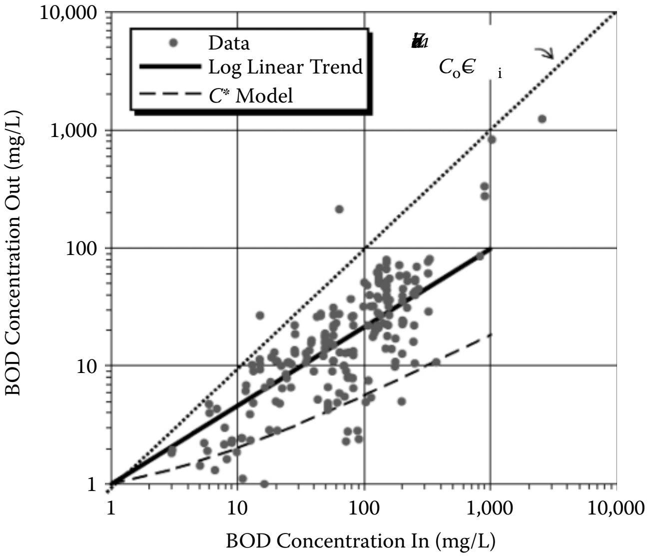


FIGURE 8.27 BOD input–output concentration graph for 202 HSSF wetlands. There is one data point per wetland, covering the entire period of record. The log-linear central tendency regression is $\log_{10}(C_o) = 0.66 \log_{10}(C_i)$, $R^2 = 0.60$. The lower bound curve, excluding 5% of the lowest values, is $C^* = 0.6 + 0.4(C_i)^{0.55}$.

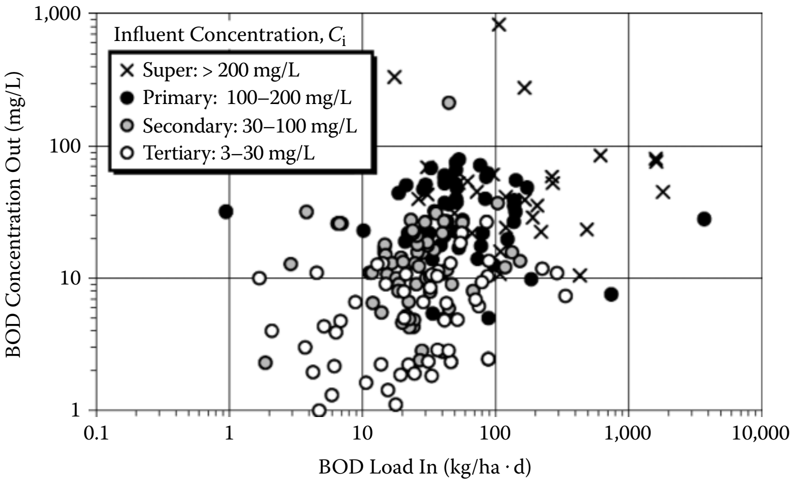


FIGURE 8.26 BOD loading graph for 202 HSSF wetlands. There is one data point per wetland, covering the entire period of record. The ranges of inlet concentrations are separated into four groups, corresponding to tertiary ($3 < C_i < 30$ mg/L) up to super ($C_i > 200$ mg/L). A slight increasing trend effluent BOD with increased BOD loading is obscured by a very large scatter.

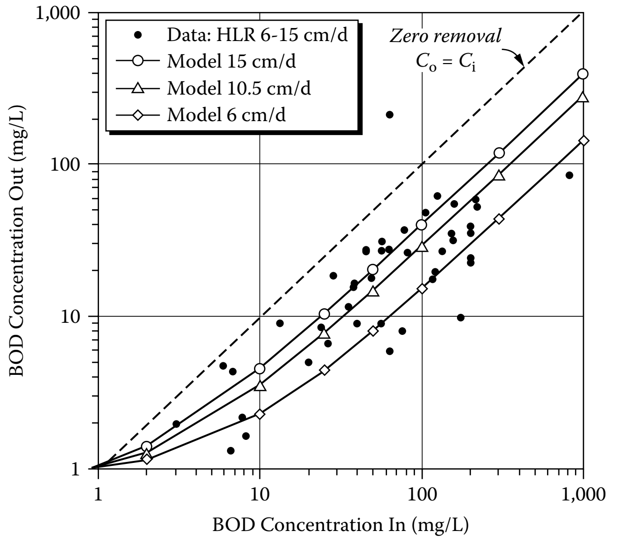


FIGURE 8.28 BOD input–output concentrations graph for HSSF wetlands with hydraulic loading rates between 6 and 15 cm/d. There is one data point per wetland, covering the entire period of record.

R^2 = 0.60. Also shown on this plot is a lower bound curve, excluding about 5% of the points as potential outliers. This bounding curve may be taken as an estimate of C^*, and is represented by:

$$C^* = 0.6 + 0.4C_i^{0.55} \tag{8.27}$$

Model Curves

A subset of the data cloud in Figure 8.26 has been reproduced in Figure 8.28, together with the *P*-*k*-C^* model results for parameter values $P = 3$, $k = 60$ m/yr, and $C^* = 1$ mg/L. The hydraulic loading is an independent parameter in that model, and the subset chosen for illustration is selected as those systems with 6 < HLR < 15 cm/d. It is seen that the model results are representative of the intersystem behavior.

As for FWS wetlands, the central messages of this modeling effort are that (1) the *P*-*k*-C^* model spans the intersystem data (as it should), but that (2) there is no resolution of the wide range of parameter values that might be selected.

Temperature Effects

The modified Arrhenius relationship is commonly used to adjust the removal rate coefficient for temperature in traditional wastewater treatment processes:

$$k = k_{20}\theta^{(T-20)} \tag{8.28}$$

where

k = rate constant at temperature T, m/yr
k_{20} = rate constant at 20°C, m/yr
T = water temperature,°C
θ = modified Arrhenius temperature factor, dimensionless

The treatment wetland literature is replete with the assertion that a θ-value of about 1.06 applies to HSSF wetlands (U.S. EPA, 1993c: 1.06; Cooper *et al*., 1996: 1.10; Water Environment Federation, 2001: 1.06; Crites *et al*., 2006: 1.06). These reports all refer to the plug flow model with $C^* = 0$. However, Kadlec and Knight (1996) could not find a temperature dependence in HSSF wetland BOD data.

Here the temperature effect on performance of several HSSF wetland systems has been analyzed with the *P*-*k*-C^* model, with $P = 3$ (Table 8.9). θ-Values range from 0.891 to 1.140, with a median of 0.981. The distribution of θ-values is given in Table 8.10. θ-Values less than unity mean slightly worse performance at higher temperatures. It is clear that the presumptive value of 1.06 is at the extreme end of the distribution, and should not be expected to occur in practice, except on rare occasions. Indeed, some researchers have concluded that there is little or no temperature effect on BOD removal in HSSF wetlands (Brix, 1998). Another feature of some existing literature is a lack of discussion of temperature effects on BOD removal in HSSF wetlands (U.S. EPA, 2000a; Wallace and Knight, 2006). The preponderance of evidence suggests that wetland BOD removal is not improved at higher wetland water temperatures.

For most HSSF systems, little or no variance is removed by adding a θ-factor to the model. It is possible that the C^*-values for a given wetland may be temperature-dependent. Decomposition of solids in the wetland may accelerate at higher temperatures, thus providing a greater BOD return rate from wetland solids. This in turn implies that background BOD could be higher in warm periods. If, as is apparently frequently the case, the wetland outlet BOD concentration is related strongly to the C^* background, then outlet BOD could be higher in summer than in winter. A first-order model without a background would show this as a reduced removal in summer. Stein *et al.* (2006b) calibrated the *k*-C^* model for batch operation ($P = \infty$), and allowed a temperature coefficient for both k and C^*. The temperature coefficients for COD C^* were found to be $0.958 < \theta < 1.029$, and thus did not resolve the issue.

Oxygen Supply

If removal of BOD is via heterotrophic oxidation of carbon compounds, oxygen transfer must be adequate to justify the first-order approximation. However, anaerobic processes can also influence BOD removal, especially in heavily loaded systems. As detailed earlier, fermentation, nitrate, iron, and sulfate reduction are all potential consumers of carbon compounds in the absence of free oxygen. Ultimately, under very low redox conditions, methanogenesis may take place. The implied maximum oxygen supply for BOD removal is simply the load of BOD removed. The systems that form the basis for Figures 8.26 and 8.27 have the median oxygen requirements shown in Table 8.11. The supply to the water in HSSF wetlands is likely to be no more than 2–4 g/m²·d (see Chapter 5). Therefore, as the incoming BOD increases to the levels seen in primary and super treatment situations,

TABLE 8.9
Arrhenius Temperature Factors for HSSF Wetlands

System	Reference	Vegetation Type	T range (°C)	Mean *C* in (mg/L)	Mean *C* out (mg/L)	Mean HLR (cm/d)	θ
Richmond, New South Wales	Bavor *et al.* (1988)	Gravel only	11–24	52	4.3	3.8	0.961
Richmond, New South Wales	Bavor *et al.* (1988)	*Typha*	11–24	52	4.7	4.6	0.960
Richmond, New South Wales	Bavor *et al.* (1988)	*Schoenoplectus*	11–24	52	5.8	5.1	0.975
Richmond, New South Wales	Bavor *et al.* (1988)	Mixed A	11–24	52	4.3	4.6	1.024
Richmond, New South Wales	Bavor *et al.* (1988)	Mixed B	11–24	52	4.6	3.8	0.985
Manhattan, Kansas	He & Mankin (2002)	PFL	5–27	178	21	13.7	0.976
Manhattan, Kansas	He & Mankin (2002)	PCL	5–27	178	58	15.3	1.018
Manhattan, Kansas	He & Mankin (2002)	PCR	5–27	178	12	11.4	1.001
Manhattan, Kansas	He & Mankin (2002)	UFL	5–27	178	27	14.7	1.019
Manhattan, Kansas	He & Mankin (2002)	UCL	5–27	178	69	13.6	1.028
Manhattan, Kansas	He & Mankin (2002)	UCR	5–27	178	31	17.1	1.023
Benton, Kentucky	TVA unpublished	3	5–25	25	8	8.4	0.921
Hardin, Kentucky	TVA unpublished	1	6–27	55	10	9.7	0.924
Grand Lake, Minnesota	Unpublished data	1	1–17	183	67	1.0	1.140
NERCC, Minnesota	Unpublished data	1 + 2	1–16	239	22	1.4	1.056
Portland, New Zealand	Unpublished data	1	11–21	30	10	5.2	0.936
Waipoua, New Zealand	Unpublished data	1	11–21	64	11	0.4	0.936
North Yorkshire 1, England	CWA (2006)	1	4–15	191	58	4.5	1.073
Cumbria, England	CWA (2006)	1	4–17	9	2	15.6	0.983
Lake Capri, Missouri	Regmi *et al.* (2003)	Nonvegetated	2–24	126	31	2.3	1.048
Lake Capri, Missouri	Regmi *et al.* (2003)	Vegetated	2–24	126	24	2.3	1.064
Fife, Scotland	CWA (2006)	1	5–16	201	35	10.3	0.993
Fife, Scotland	CWA (2006)	2	4–15	201	24	6.0	0.978
Fife, Scotland	CWA (2006)	3	4–15	201	23	11.0	0.991
Fife, Scotland	CWA (2006)	4	4–15	201	39	8.3	1.002
Hamilton, New Zealand	Tanner *et al.* (1998b)	L1	10–25	193	62	1.5	1.039
Hamilton, New Zealand	Tanner *et al.* (1998b)	L2	10–25	193	73	2.5	0.896
Hamilton, New Zealand	Tanner *et al.* (1998b)	L3	10–25	193	84	3.3	0.891
Hamilton, New Zealand	Tanner *et al.* (1998b)	L4	10–25	193	100	4.9	0.947
Hamilton, New Zealand	Tanner *et al.* (1998b)	L5	10–25	193	113	6.9	0.909
Bozeman, Montana	Stein *et al.* (2006a)	*Carex*	4–24	385	COD	Batch	0.954
Bozeman, Montana	Stein *et al.* (2006a)	*Schoenoplectus*	4–24	385	COD	Batch	0.965
Bozeman, Montana	Stein *et al.* (2006a)	*Typha*	4–24	385	COD	Batch	0.956
Bozeman, Montana	Stein *et al.* (2006a)	Control	4–24	385	COD	Batch	0.943

Note: Site names for U.K. systems are approximate.

TABLE 8.10
Percentile Points of the Distribution of Arrhenius Temperature Factors for HSSF Wetlands, Based on Table 8.9

Percentile	θ
0.05	0.904
0.10	0.922
0.20	0.940
0.30	0.956
0.40	0.967
0.50	0.981
0.60	0.993
0.70	1.018
0.80	1.026
0.90	1.054
0.95	1.067

it is unlikely that oxidative processes are entirely responsible for the destruction of BOD compounds. Additional mechanisms, such as anaerobic digestion (methanogenesis) become important contributors to removal. This lack of adequate oxygen may be overcome by (1) resorting to vertical intermittent flow wetlands, or (2) adding aeration to the HSSF bed. Vertical flow is the subject of the next section.

It is possible to design SSF wetlands that do not rely on passive diffusional processes to transfer oxygen. These systems typically operate on principles of fill-and-drain (tidal flow) (Behrends, 1999a; Austin *et al.*, 2002), or HSSF wetlands that are mechanically aerated (Dufay, 2000; Wallace, 2001; Flowers, 2002; Wallace and Lambrecht, 2003), and are proprietary (patented) systems in the United States and Canada.

TABLE 8.11
Load Reduction of BOD_5 in HSSF Wetlands

C_i =	Tertiary (g/m²·d) 3–30 mg/L	Secondary (g/m²·d) 30–100 mg/L	Primary (g/m²·d) 100–200 mg/L	Super (g/m²·d) >200 mg/L
Percentile				
0.05	0.16	0.31	1.44	2.12
0.10	0.21	0.92	1.80	2.26
0.20	0.49	1.11	2.27	4.23
0.30	0.82	1.34	2.52	5.85
0.40	1.21	1.63	2.98	10.03
0.50	1.55	1.79	3.46	10.60
0.60	2.02	2.04	3.74	16.33
0.70	2.81	2.25	4.65	19.60
0.80	3.17	2.93	6.70	42.11
0.90	3.79	3.96	10.52	76.79
0.95	7.39	6.09	12.86	122.76

Note: These amounts are the implied oxygen requirement for aerobic destruction of the compounds that comprise BOD_5.

As these systems incorporate active aeration into the wetland, it is possible to design wetlands that utilize aerobic degradation of BOD exclusively, commensurate with higher k rates. For instance, volumetric k rates for degradation of BOD in propylene glycol runoff (generated from aircraft deicing) has been demonstrated to be approximately 10–30 times higher in aerated HSSF wetlands when compared to nonaerated HSSF wetlands (Wallace *et al.*, 2007a). Design of aerated HSSF wetlands is discussed in more detail in Part II of this book.

Seasonal Trends

There are typically gentle annual cycles in the effluent BOD from HSSF wetlands (Figure 8.29). The trend is described by:

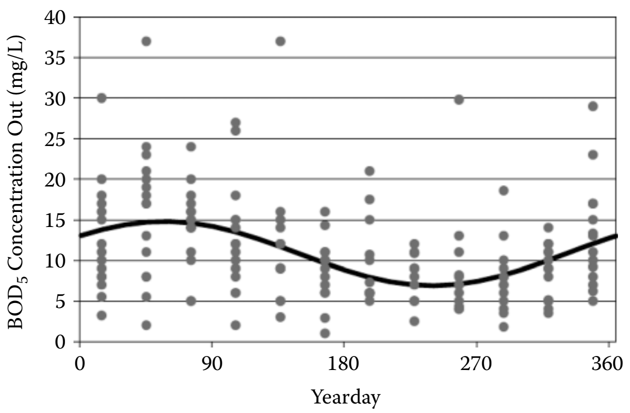


FIGURE 8.29 The annual cycle in effluent BOD_5 for the Pocahontas, Arkansas, HSSF wetland. The period of record is 14 years. (Data from WERF database (2006) *Small-Scale Constructed Wetland Treatment Systems Database* (*Project-01 CTS-5;* Final Report by Wallace and Knight, 2006). Compiled by J. Nivala and R. Clarke. Water Environment Research Foundation (WERF): Alexandria, Virginia.)

$$C = C_{avg}\left[1 + A \cdot \cos\left(\omega(t - t_{max})\right)\right] \tag{8.29}$$

where

A = trend fractional amplitude, dimensionless
C = concentration, mg/L
C_{avg} = mean annual concentration, mg/L
t = yearday, d
t_{max} = yearday for maximum concentration, d
ω = annual period, 0.01721 d^{-1}

The maximum may be at any time of the year (Table 8.12). The mean fractional amplitude is 35% of the mean.

Variability around Seasonal Trends

The considerable scatter in effluent concentrations contributes to low R^2-values for the trend lines. This behavior is of concern in wetland sizing, if the peak values of the concentrations are of importance in the permit for the project. Because stochastic behavior is present in moderate amounts, it is necessary to quantify performance variability, and ultimately to modify sizing based upon that understanding. Therefore, excursion frequencies are shown in Table 8.13.

Effects of Design and Operating Conditions

Water Depth

Bed depth (water depth) is a design variable for HSSF wetlands. As the depth is increased, the root zone changes from occupying the entire depth to occupying only the upper portion of the water column. Rooting depths are variable, but in general, roots are observed to penetrate only about 30–40 cm into HSSF beds (see Chapter 3). Deep beds will, therefore, contain a zone under the roots in which there are neither profits from root chemical effects, nor penalties from root hydraulic

TABLE 8.12
Sinusoidal Annual Trends in Effluent BOD_5 for HSSF Treatment Wetlands

Site	POR (years)	Frequency	Trend Mean (mg/L)	Trend Fractional Amplitude	Trend t_{max} (Julian day)	Trend (R^2)
Cumbria, England	3.5	Weekly	2.2	0.07	347	0.01
Leicestershire 2, England	4.5	Weekly	2.8	0.14	153	0.02
Staffordshire 3, England	3.9	Weekly	4.5	0.58	210	0.08
Fish-Royer, Indiana	2.1	Monthly	3.1	0.38	350	0.13
Haughton, Louisiana	2.0	Monthly	6.5	0.54	1	0.29
Calahan, Colorado	4.1	Monthly	10.4	0.80	61	0.25
Pocahontas, Arkansas	14.3	Monthly	10.8	0.36	58	0.15
Waipoua, New Zealand	3.6	Monthly	11.1	0.43	185	0.17
Judsonia, Arkansas	14.3	Monthly	11.5	0.28	64	0.17
Clarendon, Arkansas	14.3	Monthly	11.7	0.04	197	0.00
Eudora, Arkansas	14.3	Monthly	16.7	0.13	326	0.02
Dierks, Arkansas	5.0	Monthly	18.7	0.13	255	0.05
Lewisville, Arkansas	7.0	Monthly	20.1	0.32	292	0.10
Monterrey, Virginia	2.3	Monthly	20.6	0.20	215	0.22
Fife, Scotland (Cell 3)	1.7	Monthly	23.0	0.40	131	0.18
Las Animas, Colorado	4.0	Monthly	23.5	0.17	359	0.03
Fife, Scotland (Cell 2)	1.7	Monthly	24.8	0.39	176	0.10
North Yorkshire 2, England	6.5	Monthly	31.6	0.35	118	0.19
Valleyfield 1, United Kingdom	1.7	Monthly	37.8	0.54	122	0.30
Fife, Scotland (Cell 1)	1.7	Monthly	39.7	0.35	134	0.20
North Yorkshire 1, England	7.9	Monthly	55.4	0.22	287	0.61
Mean				**0.35**		**0.17**

Note: POR = period of record. All systems operate year-round. Site names for U.K. systems are approximate.

blockage. As in the case of FWS wetlands, increases in depth provide more detention time without adding area (footprint), but do not lower the hydraulic loading rate (see Chapter 6). The intuition of the designer is strongly influenced by the presumed form of the first-order model. If it is written for detention time, using a volumetric rate constant k_V (Equation 8.21), then it seems logical that a deeper bed provides more detention and is therefore preferable. If it is written for hydraulic loading rate, using an areal rate constant k (Equation 8.21), then the conclusion is invited that depth does not matter.

The issue may be somewhat elucidated by examining the results of two side-by-side studies of HSSF wetlands, both of which used a form of replication. The studies at Baxter, Tennessee, utilized 14 wetlands, 7 operated at 30 cm and 7 at 46 cm. These gravel cells were vegetated with bulrushes (*Scirpus validus*), and operated for three years, in two different modes (George *et al*., 1994; Kemp and George, 1997; George *et al*., 1998). In Mode 1, all were operated in parallel at different loading rates. In Mode 2, there was series and parallel operation, with recycle. Tracer testing showed approximately *N*TIS = 4, and inlet BOD_5 was 40 – 60 mg/L. The studies near Barcelona, Spain (García *et al*., 2004a), involved eight wetlands—six operated at 50-cm and two at 27-cm depth. Tracer tests showed approximately *N*TIS = 4, and inlet BOD_5 was 40 mg/L. Values of k and k_V were determined for P = 3 and C^* = 3 mg/L for both studies (Table 8.14). It is seen that both k and k_V are lower at deeper depth, meaning that deeper beds perform much more poorly than shallow. The effect is larger for k_V, which shows decreases of up to a factor of four. However, the areal k-values are also smaller at deeper depth. Thus, no matter which model is used, deeper beds are not as effective. It is apparently of no use to increase detention time by deepening the bed. These studies do not permit determination of a lower limit on bed depth, i.e., how shallow should the bed be. Coleman *et al.* (2001) also compared shallow (45 cm) and deep (60 cm) beds, and found no difference in performance at the same hydraulic loading rate, thus emphasizing that the extra depth provided no benefit.

Media Size

The concept of the HSSF wetland as a horizontal trickling filter invites the viewpoint that microbial biofilms on the media are responsible for the reduction in BOD. Small size media have greater surface area, about 150 m^2/m^3 for 25-mm spheres, and 360 m^2/m^3 for 10-mm spheres. If biofilms do the work, and if they coat the media, then a factor of two improvement would be expected for the smaller media (Khatiwada and Polprasert, 1999a). However, the evidence that such a

TABLE 8.13
Trend Multipliers for Effluent BOD Distributions in HSSF Wetlands

Site	Trend Mean (mg/L)	Trend Multiplier (80th percentile)	Trend Multiplier (90th percentile)	Trend Multiplier (95th percentile)	Trend Multiplier (99th percentile)
Cumbria, England	2.2	1.26	1.85	2.80	6.96
Leicestershire 2, England	2.8	1.50	1.97	2.66	3.51
Staffordshire 3, England	4.5	1.21	1.30	1.38	1.52
Fish-Royer, Indiana	3.1	1.28	1.72	2.96	4.48
Haughton, Louisiana	6.5	1.46	1.75	1.92	2.84
Calahan, Colorado	10.4	1.57	1.87	2.14	2.28
Pocahontas, Arkansas	10.8	1.69	2.11	2.85	3.22
Waipoua, New Zealand	11.1	1.50	1.69	2.31	2.92
Judsonia, Arkansas	11.5	1.35	1.77	1.85	2.64
Clarendon, Arkansas	11.7	1.35	1.57	1.84	2.73
Eudora, Arkansas	16.7	1.47	1.97	2.19	2.53
Dierks, Arkansas	18.7	1.54	2.07	2.46	2.67
Lewisville, Arkansas	20.1	1.65	1.80	2.07	3.47
Monterrey, Virginia	20.6	1.49	2.14	2.66	3.46
Fife, Scotland (cell 3)	23.0	1.37	1.56	1.74	1.86
Las Animas, Colorado	23.5	1.67	1.95	2.50	3.40
Fife, Scotland (cell 2)	24.8	1.26	1.63	1.94	3.10
North Yorkshire 2, England	31.6	1.30	1.60	2.11	3.50
Fife, Scotland (cell 1)	37.8	1.44	1.71	2.10	2.49
Fife, Scotland (cell 4)	39.7	1.34	1.86	2.54	3.78
North Yorkshire 1, England	55.4	1.58	1.69	1.79	1.90
Mean		**1.45**	**1.78**	**2.18**	**2.88**
SD		**0.14**	**0.21**	**0.41**	**0.73**

Note: For illustration, a trend multiplier of 1.58 for the 80th percentile for North Yorkshire 1 means that one time out of five, the effluent BOD_5 will be more than 58% higher than the trend value at that time of year. On average, the outlet BOD_5 at North Yorkshire 1 is 55.4 mg/L, and hence one sample in five may exceed 88 mg/L. Trend multiplier is $(1 + \Psi)$; see Equation 6.61. Site names for U.K. systems are approximate.

marked effect of media size is sparse, and thus the simplistic trickling filter concept should not be carried too far.

García *et al.* (2004a) performed side-by-side analyses of two media sizes (3.5 and 10 mm) in eight wetlands, using different paired depths and aspect ratios. The *k*-values display no marked effect of media size for any of the pairs, with no consistent pattern of the fine media doing better or worse. Conversely, He and Mankin (2002) found that fine media (19 mm) outperformed coarse media (38 mm). The *k*-values for *Typha* systems were 116 and 68 m/yr, and for unplanted systems were 108 and 52 m/yr for small and large media, respectively.

The selection of media size is also controlled by the need for adequate hydraulic conductivity, and by the clogging characteristics of the bed and water to be treated. Beds of sand or soil typically cannot carry the water in the subsurface mode, and then it is a moot point as to whether the finer particle size is better or not.

Vegetation

Plants provide a number of useful functions in treatment wetlands, including the possibility of oxygen release from roots

TABLE 8.14
Effect of Depth on Rate Constants for BOD_5 in HSSF Wetlands (see text for sources)

	Baxter Mode 1		Baxter Mode 2		Barcelona	
	Shallow	Deep	Shallow	Deep	Shallow	Deep
Depth (cm)	30	46	30	46	27	50
k (m/yr)	45.0	23.3	18.5	16.79	25.0	11.7
k_V (1/d)	1.027	0.347	0.422	0.256	0.635	0.160

and an increase in the sites available for bacteria (Brix, 1994b). The evidence suggests that root oxygen release is small (see Chapter 5), but other plant functions are potential contributors to improved BOD removal.

Plants are the most visible attribute of treatment wetlands, and choice of plants represents one of the few design decisions to be made. Personal preference may reflect a number of aesthetic factors. Therefore, it is not surprising that many studies have attempted to determine which plants may provide better treatment. Brisson *et al.* (2006) reviewed 27 experimental studies of which 16 assessed BOD or COD performance. They concluded: "Most studies comparing planted versus nonplanted subsurface flow constructed wetland systems for wastewater treatment show a significant and positive effect of macrophytes on pollutant removal." However, the Brisson *et al.* (2006) review utilized removal percentage as the metric, thus allowing differences in loading or detention time to cloud the issue.

One way to remove bias is via comparison of loadings and load removals, as was done by Tanner (2001b). When examined in that way, it was found that there was a slight improvement in concentration reduction for both COD and BOD—about 2–5 mg/L. The first-order areal *k*-values also remove that potential bias, and these have been computed for BOD for a sampling of side-by-side studies in Table 8.15. For simplicity, only results for *Typha* and *Schoenoplectus* are shown. Although on average the presence of plants is beneficial, that result is not unfailingly true, even in the small sample in Table 8.15. The type of plant appears to have some effect, and the Brisson *et al.* (2006) conclusion—that we should pay more attention to plant effects—seems warranted.

The European choice is most often *Phragmites*, based upon the presumption that this is the "best" plant. However, that presumption remains to be rigorously tested. Indeed, studies like that at Santee, California, Gersberg *et al.* (1986) found that bulrushes (*Schoenoplectus* spp.) were clearly superior to *Phragmites*. Theis and Young (2000) found no evidence that *Phragmites* was superior to *Schoenoplectus* in side-by-side testing.

Aspect Ratio

The Barcelona studies of García *et al.* (2004a) found no effect of aspect ratio on performance, within the range 1:1 < L:W < 2.5:1; but that range of aspect ratio is quite small. Bounds *et al.* (1998) studied three aspect ratios of L:W = 4, 10, and 30, utilizing septic tank effluent. The authors found no significant difference in percent removals due to aspect ratio, but removals were all quite high, and thus not sensitive to differences in outlet concentrations. A difference becomes apparent in terms of rate constants calculated from the Bounds *et al.* (1998) data: a 20% improvement for L:W = 30 compared to L:W = 4 during the first year, and 44% improvement during the second year. The improvements were also reflected in longitudinal profiles that decreased along the flow direction, and did not reach a plateau for any aspect ratio, but were increasingly steep as aspect ratio increased. These fractional improvements in *k*-values are commensurate with those forecast

TABLE 8.15
Comparison of First-Order Areal *k*-Values for BOD Removal for Various Plantings in SSF Wetlands

Source	Cell	*Typha* spp.	*Schoenoplectus* spp.	Gravel Only
Tanner *et al.* (1995b)	1	—	24	16
Tanner *et al.* (1995b)	2	—	19	21
Tanner *et al.*(1995b)	4	—	19	18
Regmi *et al.* (2003)	—	—	23	18
Theis and Young (2000)	—	—	16	23
Bavor *et al.* (1988)	—	61	89	98
Stein *et al.* (2006a)	—	77	79	64
Gersberg *et al.* (1986)	—	31	117	26
Heritage *et al.* (1995)	—	89	73	84
Coleman *et al.* (2001)	—	40	32	31
He and Mankin (2002)	FL	116	—	108
He and Mankin (2002)	CL	68	—	52
He and Mankin (2002)	R	124	—	119
Averages				
***Typha*/Gravel**		**76**	—	**73**
***Schoenoplectus*/Gravel**		—	**46**	**38**

Note: The model parameters used were $P = 3$ and $C^* = 3$, and period of record data was fit. The Theis and Young data are for COD, and the He and Mankin data are for COD according to a plug flow model.

for increases in P-values from about 3, up to about 10 for the highest L:W ratio. Therefore, the experimental results are in agreement with the improvements forecast from improved internal hydraulics.

8.4 BOD REMOVAL IN VF WETLANDS

Vertical flow (VF) wetlands are one of the newest forms of constructed wetlands. Typically, these systems have been dimensioned based on an empirical basis; a specified unit area (m^2) for a given organic loading (typically expressed as population equivalents, or PE). The data analysis presented in this chapter is restricted to the pulse-loaded, unsaturated downflow VF wetlands typically implemented in Europe. Other technology variants, such as tidal flow wetlands, are discussed in more detail in Part II of this book.

The operational regime of pulse-loading, followed by rest periods, allows VF systems to operate in a mode of unsaturated flow, which allows the introduction of air (and oxygen) into the VF bed (Platzer and Mauch, 1997; Cooper, 1999). Consequently, VF wetlands are more amenable to aerobic reduction of organic matter and associated BOD than HSSF wetlands.

VF wetlands have been in widespread use for over ten years now, and an increasing amount of BOD data is available to characterize the performance of these systems. This data set is largely restricted to hydraulic loadings and simple input–output relations. VF wetlands are typically operated under load-and-rest regimes, which affect system performance. At present, data on temperature and energy fluxes in VF wetlands is limited, and very few VF wetlands have been tracer tested to produce NTIS values. Due to these data limitations, the current state of the art cannot characterize VF wetland performance to the same extent as FWS and HSSF wetlands.

Graphical Relationships

The graphical display has often been adopted in the literature to characterize treatment wetland performance (Kadlec and Knight, 1996; U.S. EPA, 2000a; Wallace and Knight, 2006). Two relationships are of potential use: the concentration in–concentration out (C_i–C_o) graph and the concentration out–BOD loading (C_o–BLI) graph.

Figure 8.30 illustrates the C_i–C_o relationship for VF wetlands, based on 110 system-years of performance data. As seen in Figure 8.30, effluent concentrations (C_o) are only weakly dependent on influent concentrations (C_i), but when this data is redacted to the lowest 5% of effluent concentrations; the estimated background concentration (C^*) is 2.0 mg/L across all influent concentration ranges. A concentration of 2 mg/L is essentially the method detection limit for the BOD test, indicating that VF wetlands can be highly effective in BOD reduction.

The concentration out–load in (C_o–BLI) graph (Figure 8.31) further illustrates that treatment performance of VF wetlands is more dependent on the influent BLI. As seen in Figure 8.31,

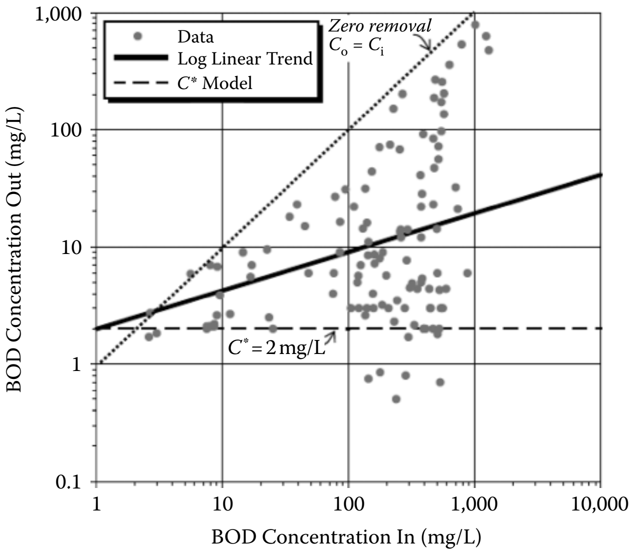


FIGURE 8.30 BOD input–output concentration graph for 62 VF wetlands. Each data point represents one system-year of performance (110 system-years total). The log-linear central trend is $\log_{10}(C_o) = 0.33\ \log_{10}(C_i) + 0.3$, $R^2 = 0.14$. The lower bound line, excluding approximately 5% of the lowest values, is $C^* = 2.0$ mg/L.

effluent concentrations (C_o) reflect a dependency of BOD influent loadings across different influent concentration ranges. The log-linear central tendency can be expressed by:

$$\log_{10}(C_o) = 0.3 + 0.71\left[\log_{10}(\text{BLI})\right] \qquad (8.30)$$
$$R^2 = 0.49$$

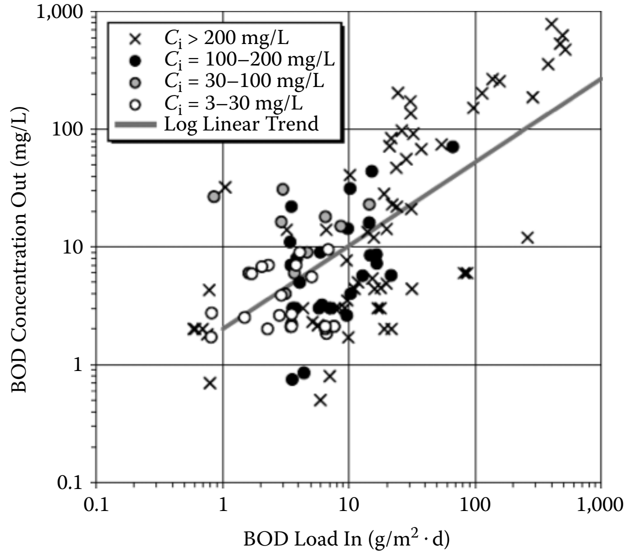


FIGURE 8.31 BOD Load In (BLI) versus BOD Concentration Out for 62 VF wetlands. Each data point is represented by one system-year of data (110 system-years total). The log-linear central trend is represented by $\log_{10}(C_o) = 0.71\ \log_{10}$ (BLI) + 0.3; $R^2 = 0.49$.

where

C_o = effluent BOD concentration, mg/L

BLI = BOD loading in, $g/m^2 \cdot d$

First-Order Modeling

In general terms, the P-k-C^* first-order model (Equation 8.20) can be used to characterize BOD removal in VF wetlands. The parameter P will always be less than the *N*TIS value due to weathering and speciation of the BOD and COD mixtures as treatment proceeds. These effects have been previously discussed in this chapter for FWS and HSSF wetlands.

A major limitation of VF wetlands is that data on *N*TIS, as determined by tracer studies, is not widely available. Data from other types of VF wetlands can be extrapolated to broadly estimate *N*TIS. For saturated upflow VF mesocosms, *N*TIS has been observed to be approximately 2 (Tanner *et al*., 2002a), and for aerated saturated downflow mesocosms, *N*TIS has been observed to be approximately 1, although a value of 2 has been proposed for full-scale systems (Wallace *et al*., 2006a). At the time of this writing, a value of $P = 2$ is postulated to represent system performance in unsaturated downflow VF wetlands. This P-value will likely change as additional data becomes available.

Based on $P = 2$ and $C^* = 2.0$ mg/L, areal k-values can be estimated on input–output data averaged over a time period sufficiently long enough to minimize seasonal changes and stochastic events. Data from 62 VF wetlands, representing 110 system-years, is summarized in Table 8.16.

By examining multiple data sets which provide a distribution of k-values, effects of BOD removal mechanisms can be examined. At low influent loadings and inlet concentrations, degradation of BOD is dominated by aerobic mechanisms. As loadings increase, aerobic processes become less probable, and anaerobic mechanisms are more likely to occur. Finally, BOD is a lumped parameter, and includes both soluble and particulate forms. If organic matter can be segregated on the surface of the wetland bed (see Figure 7.31), this will also contribute to BOD removal at high influent concentrations.

Removal as a Function of Bed Depth

Detailed transects of BOD removal as a function of bed depth are not currently available for a variety of VF wetlands. However, it has been demonstrated that the majority of microbial biomass is located in the top 20 cm of the VF bed (Langergraber *et al*., 2006b); therefore, it is highly likely that organics removal occurs preferentially in this upper region, due to filtration of particulate organic matter, greater availability of oxygen, and greater microbial biomass.

Loading Effect on *k*-Values

The plateau effect of the background concentration (C^*), affects the apparent k-values observed from input–output relationships. For VF wetlands, the area used to calculate k is normal to the flow direction; so, no underestimate of k is induced by the wetland area as the entire bed area is used for reduction of organics and BOD. However, due to the load-and-rest operational regime employed with most VF wetlands, these systems can accept very high hydraulic loading rates if loaded at influent concentrations close to C^*, leading to high estimates of k (Figure 8.32). Unlimited application of this observation leads to a design paradox; k becomes higher

TABLE 8.16
First-Order Areal Rate Constants for VF Wetlands, Based on System-Years of Performance Data

	Estimated *k*-Rates (m/yr)				
	C_i (3–30 mg/L)	C_i (30–100 mg/L)	C_i (100–200 mg/L)	C_i (>200 mg/L)	Overall
C^* (mg/L)	2.0	2.0	2.0	2.0	2.0
P	2	2	2	2	2
N	21	9	22	58	110
Percentile					
0.05	–6	11	52	31	19
0.1	21	17	62	41	34
0.2	77	32	73	47	51
0.3	160	50	110	71	78
0.4	210	57	153	101	113
0.5	471	73	187	143	146
0.6	751	91	253	160	191
0.7	1,025	101	280	246	275
0.8	1,756	113	343	316	408
0.9	2,234	116	392	554	1,006
0.95	2,402	130	473	1,165	1,694

Note: The number of data points in each concentration category is represented by N.

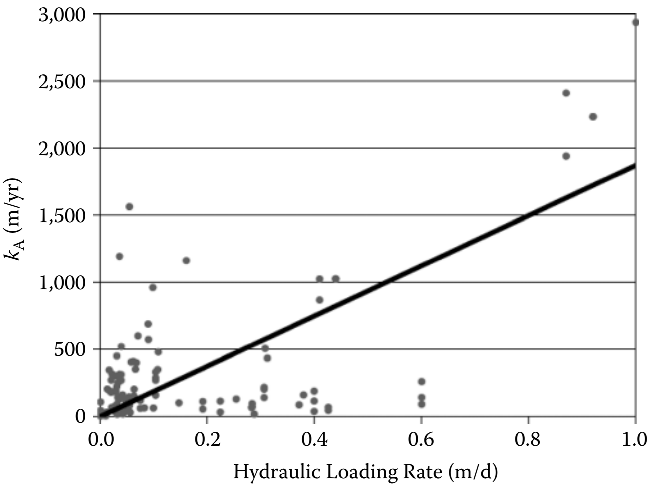


FIGURE 8.32 Dependence of the first-order areal rate constant, k_A, on hydraulic loading rate. The values $P = 2$ and $C^* = 2$ mg/L have been used. The trend line has $R^2 = 0.64$, which is obscured by a large scatter in the data, and is strongly dependent on the four datapoints at the right side of the abscissa.

with increasing loading rates, and repeated applications of this observation leads to the iterative effect of $k = \infty$, with a corresponding wetland area of zero. This is, of course, useless as a design methodology. Design of VF wetlands is addressed in more detail in Part II of this book.

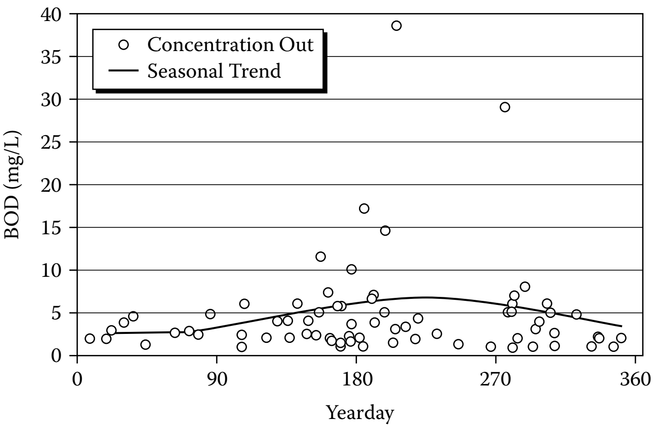


FIGURE 8.33 Example of seasonal variability in effluent concentrations; Buckinghamshire, England. Three years of performance data are represented. The mean trend is 4.6 mg/L; the fractional amplitude is 0.46, t_{max} is at 223 days, and $R^2 = 0.66$ (Data from CWA database (2006) *Constructed Wetlands Interactive Database, Version 9.02*. Compiled by G.D. Job and P.F. Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

Seasonal Effects

There are typically weak annual cycles in the effluent BOD from VF wetlands (Figure 8.33). The seasonal trend model is described in Equation 8.29. The maximum may be at any time of the year. Based on the limited data currently available, the mean fractional amplitude is approximately 35%, indicating that VF wetlands likely have seasonal changes comparable to HSSF wetlands (see Table 8.11).

At present, there is insufficient data to assess stochastic variability and temperature effects (θ-factors) for VF wetlands.

SUMMARY

Wetlands are effective in the reduction of BOD_5, as long as incoming BOD_5 exceeds the natural level at which the wetland operates. A wealth of carbon conversion processes operate in wetlands, some of which consume BOD_5, and others produce it. Both anaerobic and aerobic processes have been measured to consume carbon compounds in the wetland environment. Litter and sediment decomposition produce soluble carbon compounds. Consequently, the simplest mass balance model must include both consumption and generation of these substances.

Profiles of BOD_5 along the flow direction may be described by the *P-k-C** model. However, there are wide ranges of parameter values; so, no universal recommended values of *P*, *k*, and *C** can be developed. The background BOD_5 depends somewhat on season of the year. Neither the rate constant nor the background BOD_5 depends strongly on temperature. Because BOD_5 is a mixture, and subject to weathering, rate constants change during the removal process. This leads to *P*-values that are considerably less than determined from inert tracer testing. Variations in these parameters with temperature cannot be quantified adequately from the existing database, and consequently all variability must be absorbed into the performance spectrum.

Most operating FWS wetlands are overdesigned for BOD_5 removal, and hence effluent concentrations are at or near background levels, in the 1–10 mg/L range.

HSSF wetlands inherently oxygen-transfer limited systems; therefore, there is presumptively a shift in degradation processes (from aerobic to anaerobic) and the influent mass load increases, although this relationship is obscured by the considerable scatter in the available HSSF data set.

VF wetlands (operating on pulse-load, unsaturated flow principles) are considerably more effective in degradation of organic matter (including BOD) as the load-and-rest operating protocol for these wetlands allows the introduction of atmospheric oxygen directly into the wetland bed.

All types of treatment wetlands have seasonally variable changes in BOD effluent quality. These seasonal changes are driven by climate, plant biomass cycling, and water temperatures.

9 Nitrogen

Nitrogen compounds are among the principal constituents of concern in wastewater because of their role in eutrophication, their effect on the oxygen content of receiving waters, and their toxicity to aquatic invertebrate and vertebrate species. These compounds also augment plant growth, which in turn stimulates the biogeochemical cycles of the wetland. The wetland nitrogen cycle is very complex, and control of even the most basic chemical transformations of this element is a challenge in ecological engineering. This chapter describes the wetland nitrogen cycle, summarizes current knowledge about environmental factors that control nitrogen transformations, and provides alternative approaches that can be used to design wetland treatment systems to treat nitrogen.

9.1 NITROGEN FORMS IN WETLAND WATERS

The most important inorganic forms of nitrogen in wetlands treating municipal or domestic wastewater are ammonia (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), nitrous oxide (N_2O), and dissolved elemental nitrogen or dinitrogen gas (N_2). Nitrogen is also invariably present in FWS wetlands in organic forms. Both dissolved and particulate forms may be present, but in most cases there is little particulate nitrogen in settled wetland surface waters.

Common analytical methods include procedures for determination of total or dissolved forms (APHA, 2005). These include

- Nitrate
- Nitrite
- Ammonia
- Total Kjeldahl nitrogen (TKN) = (organic + ammonia nitrogen)

From these basic measures, several derived concentrations may be computed:

- Oxidized nitrogen = nitrate + nitrite
- Inorganic nitrogen = oxidized nitrogen + ammonia
- Organic nitrogen = TKN − ammonia
- Total nitrogen = TKN + oxidized nitrogen

Each category can be the subject of wetland effluent quality regulation, and each may represent an important feature of wetland water quality, depending upon the nature of source waters.

As treatment wetland technology develops, nondomestic source waters are of increasing interest, thus bringing attention to other nitrogen compounds. Examples include

- Polymer industry wastewaters, which contain amines (RNH_2, where R is an aliphatic hydrocarbon) (Beeman and Reitberger, 2003)
- Potato wastewaters, which contain imides (RCO–NH–OCR', where R and R' are aliphatic hydrocarbons) (Kadlec *et al*., 1997)
- Aluminum and gold processing waste leachates, which contain cyanide (CN^-) (Bishay and Kadlec, 2005; Gessner *et al*., 2005)
- Chlorinated effluents, which develop chloramines in the wetland ($NH_xCl_y^-$) (Zheng *et al*., 2004)
- Triazine pesticides in agricultural runoff (e.g., atrazine, $C_8H_{13}N_5Cl$) (Moore *et al*., 2000b)

These and other specialty applications of interest are discussed in Chapters 13 and 25.

Organic Nitrogen

Organic nitrogen is made up of a variety of compounds including amino acids, urea and uric acid, and purines and pyrimidines. Amino acids are the main components of proteins, which are a group of complex organic compounds essential to all forms of life. Amino acids consist of an amine group (–NH_2) and an acid group (–COOH) attached to the terminal carbon atom of a variety of straight carbon chain and aromatic organic compounds. Organic forms of nitrogen, primarily as amino acids, typically makes up from 1–7% of the dry weight of plants and animals.

Urea (CNH_4O) and uric acid ($C_4N_4H_4O_3$) are among the simplest forms of organic nitrogen in aquatic systems. Urea is formed by mammals as a physiological mechanism to dispose of ammonia that results when amino acids are used for energy production. Because ammonia is toxic, it must be converted to a less toxic form, urea, by the addition of carbon dioxide. Uric acid is produced by insects and birds for the same purpose. These organic forms of nitrogen are important in wetland treatment because they are readily hydrolyzed, chemically or microbially, resulting in the release of ammonia.

Pyrimidines and purines are heterocyclic organic compounds in which nitrogen replaces two or more of the carbon atoms in the aromatic ring. Pyrimidines consist of a single heterocyclic ring, and purines contain two interconnected rings. These compounds are synthesized from amino acids to become the main building blocks of the nucleotides that make up DNA in living organisms.

Wastewaters contain varying amounts of organic nitrogen, depending upon the source. Nitrogen in domestic sewage comprises about 60% ammonia and 40% organic nitrogen (U.S. EPA, 1993b). Activated sludge treatment processes typically reduce this fraction considerably, but facultative lagoon effluents may retain the same proportions while reducing total nitrogen (TN). Food processing effluents may contain very high amounts of organic nitrogen.

Ammonia

Ammonia exists in water solution as either as un-ionized ammonia (NH_3) or ionized ammonia (NH_4^+, ammonium ion), depending on water temperature and pH:

$$NH_3 + H_2O \rightleftharpoons NH_4^+ + OH^- \tag{9.1}$$

Total ammonia is equal to the sum of the un-ionized and the ionized ammonia, and is designated as *ammonia nitrogen* in this book. The fraction of un-ionized ammonia in water may be estimated from equilibrium conditions, given by

$$\log_{10} K_d = \log_{10} \frac{C_{IA}}{C_{UA}} = 0.09018 - \left(\frac{272{,}992}{T + 273.16} \right) - \text{pH} \tag{9.2}$$

where

C_{IA} = ionized ammonia concentration, mg/L
C_{UA} = unionized ammonia concentration, mg/L
K_d = dissociation constant, dimensionless
T = water temperature, °C

The ionized form is predominant in most wetland systems because of moderate pH and temperature, and is designated as ammonium nitrogen in this book. For a typical "average" environmental condition of 25°C and a pH of 7, un-ionized ammonia is only 0.6% of the total ammonia present. At a pH of 9.5 and a temperature of 30°C, the percentage of total ammonia present in the un-ionized form increases to 72%. At lower pH and temperature values, this percentage decreases significantly and presumably from wetlands under high pH and temperature conditions. Un-ionized ammonia is toxic to fish and other forms of aquatic life at low concentrations typically at concentrations >0.2 mg/L. U.S. EPA promulgates acute and chronic criteria for toxicity, and the reader is encouraged to consult the latest publication of such limits. Wetlands are useful for modulation of un-ionized ammonia, because they create circumneutral pH, and may lower water temperatures for warm effluents (Kadlec and Pries, 2004).

Ammonia typically comprises more than half of the TN in a variety of municipal and domestic effluents, where concentrations often are in the range of 20–60 mg/L. However, ammonia concentrations in food processing wastewaters treated in wetlands can exceed 100 mg/L (Van Oostrom and Cooper, 1990; Kadlec *et al.*, 1997). Landfill leachates, particularly from recently closed and capped landfills, can contain hundreds of mg/L (Bulc *et al.*, 1997; McBean and Rovers, 1999; Kadlec, 2003c).

Because ammonia is one of the principal forms of nitrogen found in many wastewaters and because of its potential role in degrading the environmental condition of wetlands and other receiving waters, reducing ammonia concentration drives the design process for many wetland treatment systems.

Oxidized Nitrogen

Nitrite (NO_2^-) is an intermediate oxidation state of nitrogen (oxidation state of +3) between ammonia (−3) and nitrate (+5). Because of this intermediate energetic condition, nitrite is not chemically stable in most wetlands and is generally found only at very low concentrations. Nitrate (NO_3^-) is the most highly oxidized form of nitrogen (oxidation state of +5) found in wetlands. Because of this oxidation state, nitrate is chemically stable and would persist unchanged if not for several energy-consuming biological nitrogen transformation processes that occur. Nitrate can serve as an essential nutrient for plant growth, but in excess, it leads to eutrophication of surface water. Nitrate and nitrite are also important in water quality control because they are potentially toxic to infants (they result in a potentially fatal condition known as methylglobanemia) when present in drinking waters derived from polluted surface or groundwater supplies. The current regulatory criteria for nitrate in groundwater and drinking water supplies in the United States is 10 mg/L.

Oxidized nitrogen is typically near zero in sewage and in secondarily treated effluents, including secondary activated sludge and facultative lagoon waters. However, nitrate may seasonally be the dominant form in nitrified secondary effluents. It is present in agricultural runoff due to the oxidation of ammonia fertilizers in the vadose zone of farm fields, and may reach 40 mg/L in some cases.

9.2 WETLAND NITROGEN STORAGES

Organic nitrogen compounds are a significant fraction of the dry weight of wetland plants, detritus, microbes, wildlife, and soils. The mass of these nitrogen storages varies in different wetland types. A general idea of the sizes of these different storage compartments is necessary to understand the nitrogen fluxes discussed in this chapter (Figure 9.1).

Soils and Sediments

The total of newly accreted organic materials at the Sacramento, California, FWS site had about 1.5% nitrogen (Nolte and Associates, 1998b). At the Houghton Lake, Michigan, and WCA2A, Florida, FWS sites, the organic sediments and soils averaged 3.13 ± 0.26 and 2.97 ± 0.37% nitrogen by dry weight, respectively. At both these sites, there was essentially no vertical profile in mass nitrogen percentage, but there was an increase in soil bulk density with depth for both. As a result, the volumetric storage of nitrogen increased with depth (Figure 9.2). The resulting

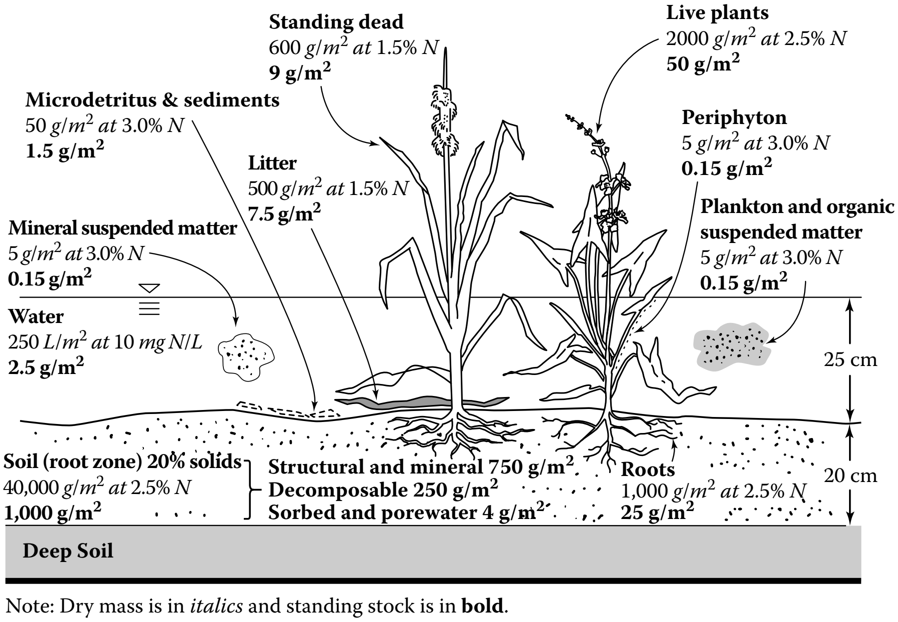


FIGURE 9.1 Nitrogen storages in a densely vegetated hypothetical FWS treatment wetland. Note that most of the stored nitrogen is in soils and sediments (≈1,000 gN/m²), second most is in plant materials (≈100 gN/m²), and least is in mobile forms in the water column (≈5 gN/m²).

nitrogen storage is about 500–2,000 gN/m² in the upper 30 cm of organic wetland sediments. For instance, the data of Figure 9.2 indicate approximately 700–800 gN/m² for Houghton Lake and WCA2A, respectively.

It is not common for the new sediments and soils in a treatment wetland to be inorganic in character. However, systems treating runoff may receive considerable quantities of inorganic solids from soil erosion in the watershed, which then combine with organic materials generated within the wetland. An example is Chiricahueto marsh in Mexico (Soto-Jiménez *et al.*, 2003). Agricultural runoff brought water at about 15 mg/L of TN to the marsh for over 50 years. The soil column is now mostly inorganic, with less than 5% carbon (Figure 9.3). Mineral matter typically has a low nitrogen content, and consequently the nitrogen percentages were low, less than 0.4% dry weight. Both carbon and nitrogen decreased together as depth increased, indicating that most of the soil nitrogen was associated with the organic content. The nitrogen content of the upper 30 cm at Chiricahueto was 330 gN/m².

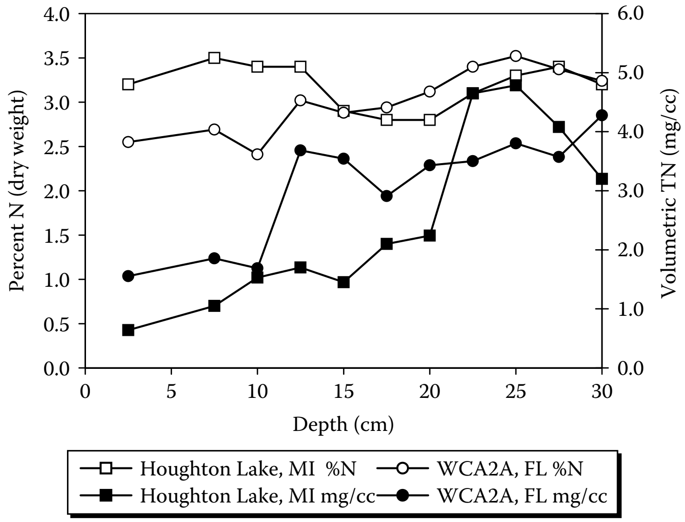


FIGURE 9.2 Vertical variation in mass and volume concentrations soil of nitrogen in two FWS treatment wetlands. Houghton Lake, Michigan, data were acquired beneath waters at about 10 mg/L TN after nine years' exposure, and WCA2A, Florida, data were acquired at a site with pore water ammonia of 1.5–3.5 mg/L, and surface water of about 2.4 mg/L total nitrogen, after about 20 years' exposure. (Data for Houghton Lake: unpublished data; data for WCA2A: unpublished data; and Reddy *et al.* (1991) *Physico-Chemical Properties of Soils in the Water Conservation Area 2 of the Everglades*. Report to the South Florida Water Management District, West Palm Beach, Florida.)

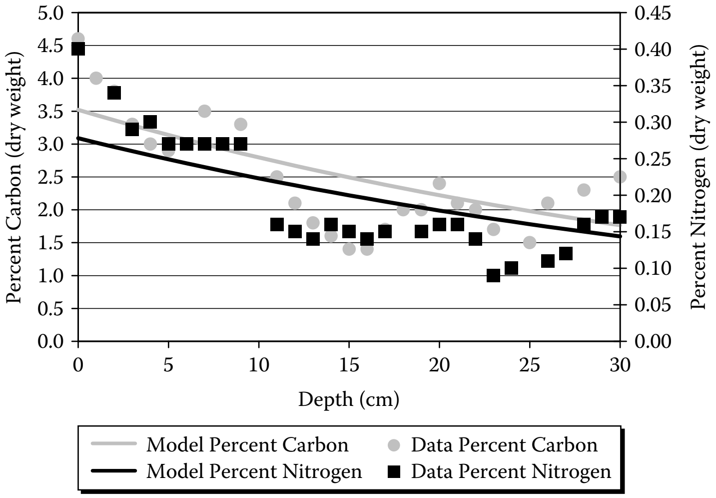


FIGURE 9.3 The decline of carbon and nitrogen with depth in a FWS wetland receiving agricultural runoff, at Chiricahueto, Mexico. (Data from Soto-Jiménez *et al.* (2003) *Water Research*, 37: 719–728.)

Biomass

The TN content of living biomass in marsh wetlands varies considerably among species, among plant parts, and among wetland sites. There is little variation from location to location within a homogeneous stand (Boyd, 1978). Example ranges of dry weight nitrogen percentages in natural wetlands are: 0.9–2.6% for emergent plants; 1.96–3.8% for floating leaved plants; and 2.4–2.9% for submersed plants (Boyd, 1978).

Treatment wetlands are often nutrient-enriched and display higher values of tissue nutrient concentrations than natural wetlands. For instance, live cattail leaves in the discharge area of the Houghton Lake, Michigan, FWS wetland averaged 2.0% N; those in nutrient-poor control areas averaged 1.1% N; dead leaves showed 1.6 versus 0.7% N, and litter leaves showed 3.6 versus 1.5% N, respectively (Table 9.1). Total biomass is enhanced by fertilization with effluent, and this compounds the effect of increased nutrient content, to produce large

TABLE 9.1
Nitrogen Content (gN/m²) of Vegetation in Treatment and Natural Areas at the Houghton Lake, Michigan, Treatment Wetland Site

	Control (DIN ≤ 0.1 mg/L)			Discharge (DIN ≈ 15 mg/L)		
	Biomass (g/m²)	Content (%)	Crop (gN/m²)	Biomass (g/m²)	Content (%)	Crop (gN/m²)
Live						
1995	368	1.08	4.0	1,086	1.98	18.9
1996	773	1.08	8.2	1,323	2.37	30.3
1997	504	1.00	5.1	1,200	2.11	25.4
1998	311	1.11	3.4	1,333	1.65	22.5
4-year mean	*489*	*1.07*	*5.2*	*1,235*	*2.03*	*24.3*
Standing Dead						
1995	642	0.69	4.8	917	1.07	10.1
1996	390	0.58	2.3	392	1.54	5.8
1997	190	0.77	1.4	1,642	2.02	32.6
1998	401	0.61	2.1	1,336	1.59	22.3
4-year mean	*406*	*0.66*	*2.7*	*1,072*	*1.55*	*17.7*
Litter						
1996	84	1.60	1.4	1,769	3.60	62.2
1997	42	1.75	0.8	2,193	3.63	79.3
1998	135	1.75	2.3	3,090	3.55	111.1
3-year mean	*87*	*1.70*	*1.5*	*2,351*	*3.59*	*84.2*
Total Above	**982**	**1.08**	**9.4**	**4,658**	**2.37**	**126**

Note: DIN = dissolved inorganic nitrogen = oxidized plus ammonia nitrogen.

Source: Unpublished data.

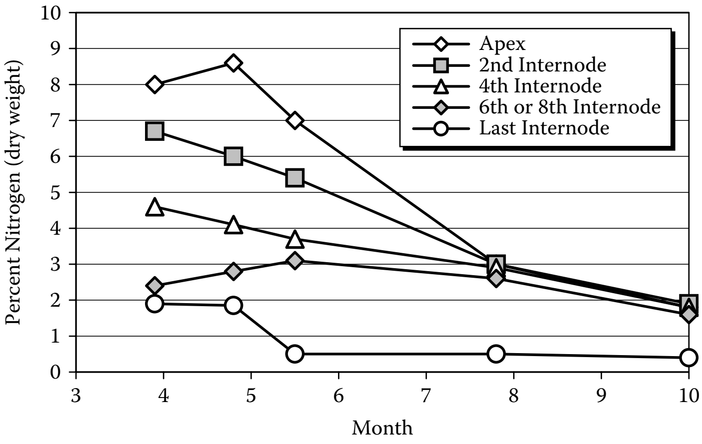


FIGURE 9.4 Nitrogen content in *Phragmites australis* as a function of season and position aboveground. The site was a highly productive reed stand, which generated 1,500 g/m^2 of biomass over the June–August period. Redrawn from the data of Kühl and Kohl (1993). (Graph from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

storages in treatment areas compared to unfertilized natural wetlands.

Different plant parts may show large differences in nitrogen content, and the seasonal variability may be very large. The extent of this variability is shown in Figure 9.4 for *Phragmites australis*, for a reed stand in the margin of Templiner See, a heavily loaded eutrophic shallow lake in Germany (Kühl and Kohl, 1993). Biomass collected at the end of the growing season displays much lower nitrogen content than in spring. Klopatek (1978) has shown trends of the same magnitude for cattail roots and shoots. It is apparent that the timing and location of vegetation samples can greatly affect subsequent calculations of nitrogen storage in biomass. The decline of aboveground tissue nutrient content is a common phenomenon in both treatment and natural wetlands (Table 9.2) and results in a markedly lower tissue nitrogen concentration at the end of the growing season. This is partly due to translocation to belowground rhizomes, which is discussed in a following section.

These seasonal storages reflect the growth cycle of the plant in question. The processes of growth, death, litterfall, and decomposition operate year-round, and with different speed and seasonality depending on climatic conditions and genotypical habit. Even in cold climates, the total annual growth is slightly larger than the end-of-season standing crop, by about 20% (Whigham *et al.*, 1978). In warm climates, measurements show 3.5–10 turnovers of the live aboveground standing crop in the course of a year (Davis, 1994). Decay and translocation processes release most of the nitrogen uptake, with the residual accreting as new sediments and soils.

TABLE 9.2
Whole Plant, Aboveground Foliar Nitrogen Concentration Declines through the Growing Season

Plant Species	Location	Water	Initial N (%)	Decline Rate (%/d)	R^2	Reference
Typha latifolia	South Carolina	N	2.47	0.0133	0.90	Boyd (1971)
Typha latifolia	Michigan	S	1.00	0.0004	0.75	Houghton Lake, Michigan, unpublished data
Typha angustifolia	Michigan	S	1.33	0.0027	0.77	Houghton Lake, Michigan, unpublished data
Typha spp.	Minnesota	N	1.80	0.0063	0.99	Pratt *et al.* (1980)
Typha spp.	Minnesota	N	1.70	0.0075	0.86	Pratt *et al.* (1980)
Scirpus validus[a]	New Zealand	P	1.46	0.0061	0.80	Tanner (2001a)
Scirpus validus	New Zealand	P	1.61	0.0059	0.82	Tanner (2001a)
Scirpus validus	New Zealand	P	1.79	0.0058	0.82	Tanner (2001a)
Scirpus validus	New Zealand	P	1.93	0.0087	0.88	Tanner (2001a)
Phragmites australis	The Netherlands	N	2.74	0.0100	0.90	Mueleman *et al.* (2002)
Phragmites australis	Australia	AR	4.22	0.0146	0.93	Hocking (1989a, b)
Phragmites australis	The Netherlands	P	2.54	0.0070	0.96	Mueleman *et al.* (2002)

Note: Initial %N is at the start of the growing season. Water type is N = no wastewater; S = nutrients at secondary treatment levels; P = nutrients at primary treatment levels; AR = agricultural runoff.

[a] Currently known as *Schoenoplectus tabernaemontani.*

TABLE 9.3
End of Season Plant Biomass in Wetlands

Species	Location	Reference	Water	S/P/E	Live Above (g/m²)	Total Above (g/m²)	Roots and Rhizomes (g/m²)
Cattails							
Typha latifolia	Wisconsin	Smith *et al.* (1988)	N	105/245/290	—	1,400	450
Typha latifolia	Texas	Hill (1987)	N	60/240/345	—	2,500	2,200
Typha glauca	Iowa	van der Valk and Davis (1978)	N	120/265/290	2,000	—	1,340
Typha latifolia	Michigan	Houghton Lake, Michigan, unpublished data	N	120/245/275	490	890	6,200
Typha latifolia	Michigan	Houghton Lake, Michigan, unpublished data	S	120/245/275	1,240	2,310	2,900
Typha latifolia	Kentucky	Pullin and Hammer (1989)	P	—	5,602	—	3,817
Typha angustifolia	Kentucky	Pullin and Hammer (1989)	P	—	5,538	—	4,860
Bulrushes							
Scirpus fluviatilis	Iowa	van der Valk and Davis (1978)	N	130/265/285	790	—	1,370
Scirpus validus[a]	Iowa	van der Valk and Davis (1978)	N	120/210/300	2,100	—	1,520
Scirpus validus	New Zealand	Tanner (2001a)	P	30/205/350	2,100	2,650	1,200
Scirpus validus	Kentucky	Pullin and Hammer (1989)	P	—	—	2,355	7,376
Scirpus cyperinus	Kentucky	Pullin and Hammer (1989)	P	—	—	3,247	12,495
Phragmites							
Phragmites australis	United Kingdom	Mason and Bryant (1975)	N	75/220/305	942	1,275	—
Phragmites australis	Iowa	van der Valk and Davis (1978)	N	—	—	1,110	1,260
Phragmites australis	The Netherlands	Mueleman *et al.* (2002)	N	105/255/350	2,900	3,200	7,150
Phragmites australis	Brisbane	Greenway (2002)	S	—	1,460	2,520	1,180
Phragmites australis	The Netherlands	Mueleman *et al.* (2002)	P	105/255/355	5,000	5,500	3,890
Phragmites australis	New York	Peverly *et al.* (1993)	L	100/270/330	10,800	—	8,700

Note: Water type is N = no wastewater; S = nutrients at secondary treatment levels; P = nutrients at primary treatment levels; L = landfill leachate with about 300 gN/m³. S/P/E refers to the start, peak, and end year-days of the growing season (182 days added for southern hemisphere).

[a] Currently known as *Schoenoplectus tabernaemontani.*

A common point of reference often used to assay biomass nitrogen is the end of the growing season. The compartments most often analyzed are live aboveground plant tissues, standing dead and litter, and belowground roots and rhizomes (Table 9.3). It is seen that a considerable fraction of the biomass is belowground, which is particularly troublesome from the standpoint of sampling, and hence often omitted. *A rough estimate of nitrogen storages in Table 9.3 may be obtained by multiplying the dry biomass by 2% nitrogen, resulting in a range of about 100–300 gN/m².* In treatment wetlands that are lightly loaded, this storage may be an important factor in the nitrogen budget, on a seasonal basis.

9.3 NITROGEN TRANSFORMATIONS IN WETLANDS

Figure 9.5 shows the principal components of the nitrogen cycle in wetlands. The various forms of nitrogen are continually involved in chemical transformations from inorganic to organic compounds and back from organic to inorganic. Some of these processes require energy (typically derived from an organic carbon source), and others release energy, which is used by organisms for growth and survival. Most of the chemical changes are controlled through the production of enzymes and catalysts by the living organisms they benefit.

The several nitrogenous chemical species are interrelated by a reaction sequence. Nitrogen is speciated in several forms in wetlands, as well as partitioned into water, sediment, and biomass phases. An FWS wetland is also stratified vertically into zones which promote different nitrogen reactions. As a further complicating factor, microenvironments around individual plant roots may differ from the bulk surroundings (Reddy and D'Angelo, 1994). Although the detailed processes are well known, they have not been adequately quantified as an integrated network for the wetland environment.

A number of processes transfer nitrogen compounds from one point to another in wetlands without resulting in a molecular transformation. These physical transfer processes include, but are not limited to the following: (1) particulate settling and resuspension, (2) diffusion of dissolved forms, (3) plant translocation, (4) litterfall, (5) ammonia volatilization, and (6) sorption of soluble nitrogen on substrates. In addition to the physical translocation of nitrogen compounds in wetlands, five principal processes transform nitrogen from one form to another: (1) ammonification (mineralization), (2) nitrification, (3) denitrification, (4) assimilation, and (5) decomposition. A detailed understanding of these nitrogen transfer and transformation processes is important for understanding wetland treatment systems. The sections below describe these processes and the environmental factors that

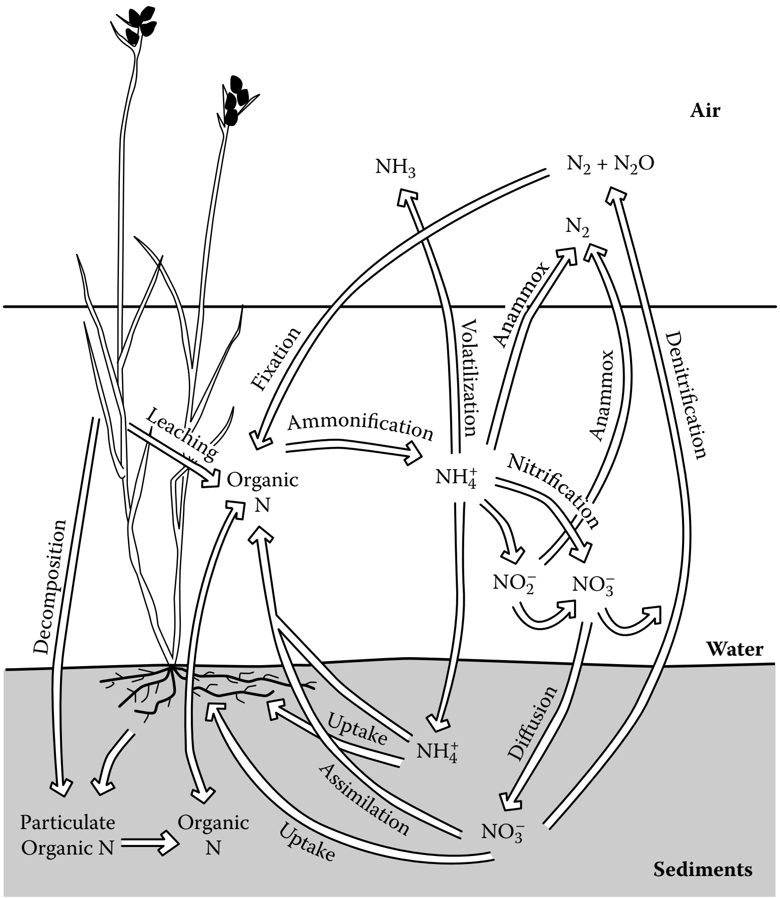


FIGURE 9.5 Simplified nitrogen cycle for a FWS treatment wetland. (Modified from Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

regulate the transformations. Later in this chapter, empirical and theoretical design methods are presented for predicting the treatment wetland area necessary to accomplish the given nitrogen transformations.

Physical Processes

The wetland nitrogen cycle includes a number of pathways that do not result in a molecular transformation of the affected nitrogen compound. These physical processes include atmospheric nitrogen inputs, ammonia adsorption, and ammonia volatilization. Sedimentation may also remove particulate nitrogen from the water, either as a structural component of the total suspended solids (TSS), or as sorbed ammonia (see Chapter 7).

Atmospheric Deposition

Atmospheric deposition of nitrogen contributes measurable quantities of nitrogen to receiving land areas. All forms are involved: particulate and dissolved, and inorganic and organic. Wetfall contributes more than dryfall, and rain contributes more than snow (Table 9.4). The nitrogen concentration of rainfall is highly variable depending on atmospheric conditions, air pollution, and geographical location. A typical range of TN concentrations associated with rainfall is 0.5–3.0 mg/L, with more than half of this present as ammonia and nitrate nitrogen.

Some dryfall of nitrogen is also from deposition of organic dust containing organic and ammonia nitrogen. Typical dryfall nitrogen inputs are less than wetfall amounts. These concentrations can be used with local rainfall amounts to estimate rainfall inputs in nitrogen mass balances (Table 9.4). Annual total atmospheric nitrogen loadings are 10–20 kg/ha·yr. Consequently, atmospheric sources are almost always a negligible contribution to the wetland nitrogen budget for all but ombrotrophic, nontreatment wetlands.

Ammonia Sorption

Oxidized nitrogen forms (e.g., nitrite and nitrate) do not bind to solid substrates, but ammonia is capable of sorption to both organic and inorganic substrates. Because of the positive charge on the ammonium ion, it is subject to cation exchange. Ionized ammonia may therefore be removed from water through exchange with detritus and inorganic sediments in FWS wetlands, or the media in SSF wetlands. The adsorbed ammonia is bound loosely to the substrate and can be released easily when water chemistry conditions change.

TABLE 9.4
Atmospheric Deposition of Nitrogen

Location and Nitrogen Form		Type of Deposition	Estimated Precipitation (mm)	Concentration (mg/L)	Load (kg/ha·yr)	Reference
Geneva, New York	Inorganic	Wet + dry	993	1.1	10.9	U.S. EPA (1993b)
Coshocton, Ohio	Inorganic	Wet + dry	939	0.8	7.5	U.S. EPA (1993b)
	Organic	Wet + dry	—	0.37	3.5	
Cincinnati, Ohio	Inorganic	Wet + dry	1,020	0.69	7.0	U.S. EPA (1993b)
	Organic	Wet + dry	—	0.58	5.9	
Seattle, Washington	Nitrate	Dry	—	—	0.7	U.S. EPA (1993b)
Ottawa, Ontario	Inorganic	Snow	147	0.85	1.3	U.S. EPA (1993b)
	Nitrate	Rain	724	0.35	15.6	
	Ammonia	Rain	—	1.8	13.0	
Hamilton, Ontario	Total nitrogen	Wet	818	0.49	4	U.S. EPA (1993b)
	Total nitrogen	Dry	—	—	2.5	
Southern Florida	Inorganic	Wet + dry	1,500	0.75	6.1	South Florida Water Management District, unpublished data
	Organic	Wet + dry	—	1.13	9.3	
	Particulate	Wet + dry	—	0.94	7.7	
Midwestern United States	Ammonia	Wet + dry	889	0.34–0.45	3–4	U.S. EPA (2001b)
North Carolina	Nitrate	Wet	1,355	0.25	3.4	Whitall and Paerl (2001)
	Ammonia	Wet	—	0.23	3.1	
	Organic	Wet	—	0.23	3.2	
Chesapeake Bay		Wet (2/3)				Sheeder *et al.* (2002)
	Inorganic	Dry (1/3)	1,143	0.34–1.62	4–19	
Southern Sweden	Total nitrogen	Wet + dry	569	2.6–4.4	15–25	U.S. EPA (1993b)
Central Europe	Total nitrogen	Wet + dry	866	2.3–3.5	20–30	U.S. EPA (1993b)

At a given ammonia concentration in the water column, a fixed amount of ammonia is adsorbed to and saturates the available attachment sites.

The character of the substrate is an important determinant of the amount of sorption or exchange (Figure 9.6). Natural zeolites have more exchange capacity than do the gravels usually employed in SSF wetlands, by more than a factor of 100. Organic sediments and peats in FWS wetlands have capacities intermediate to zeolites and gravels. The exchange reaction involves protons on the substrate and ammonia:

$$RH^+ + NH_4^+ + OH^- \rightleftharpoons RNH_4^+ + H_2O \quad (9.3)$$

where R represents a ligand, such as the humic substances found in peat. Other cations, including sodium (Na^+), calcium (Ca^{2+}) and magnesium (Mg^{2+}), compete for exchange sites, and reduce the potential for ammonia exchange (Weatherly and Miladinovic, 2004). Hydrogen ions are also important, because these too reduce the exchange capacity. For example, McNevin and Barford (2001) found the direct dependence for Killarney peat, over the range 3.9 < pH < 7.5 to follow:

$$K_{exch} = \frac{C_S}{C_L} = 0.0018(\text{pH})^{5.438} \quad (9.4)$$

where

C_L = ammonia concentration in water, mg/L
C_S = ammonia concentration on solid, mg/kg
K_{exch} = partition coefficient, L/kg

When the ammonia concentration in the water column is reduced, some ammonia will be desorbed to regain equilibrium

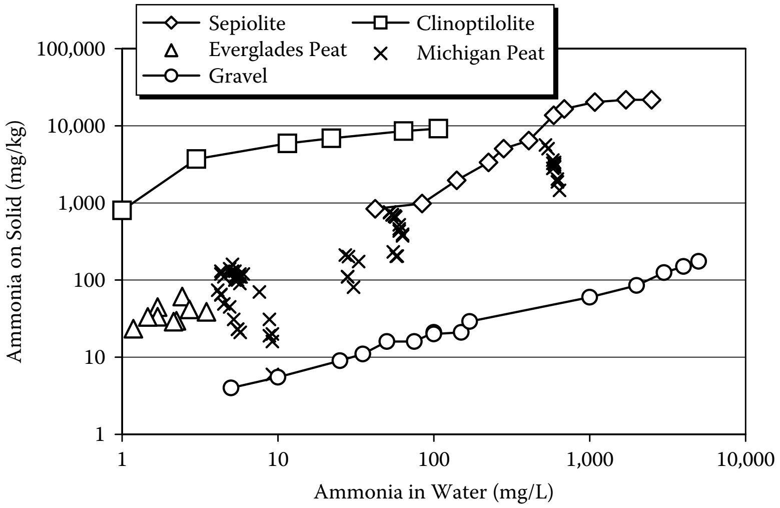


FIGURE 9.6 Ammonium adsorption on FWS and SSF wetland substrates. (The gravel data are from Sikora *et al.* (1994) *Ammonium and phosphorus removal in constructed wetlands with recirculating subsurface flow: Removal rates and mechanisms*. Jiang (Ed.), Proceedings of the 4th International Conference on Wetland Systems for Water Pollution Control, 6–10 November 1994; IWA: Guangzhou, P.R. China, pp. 147–161. Everglades peat data from Reddy *et al.* (1991) *Physico-Chemical Properties of Soils in the Water Conservation Area 2 of the Everglades*. Report to the South Florida Water Management District, West Palm Beach, Florida. Michigan peat data from unpublished results at Houghton Lake. Sepiolite data from Balci (2004) *Water Research* 38(5): 1129–1138. Clinoptilolite data from Weatherly and Miladinovic (2004) *Water Research* 38(20): 4305–4312.)

with the new concentration. If the ammonia concentration in the water column is increased, the adsorbed ammonia will also increase.

The mass of sorbed ammonia nitrogen on detritus and sediment in an FWS wetland is not large, and is very labile. The top 20 cm of the wetland substrate may contain up to 20 gN/m^2 in exchangeable form for a peat exposed to 10 mg/L ammonium nitrogen. This pool of nitrogen is quickly established at moderate nitrogen loadings (see Chapter 10 for an analogous discussion of sorption saturation times for phosphorus). At light nitrogen loadings, a short start-up period may be influenced by this storage.

Wittgren and Maehlum (1997) suggest that seasonal sorption could store ammonia for later use and release. Riley *et al.* (2005) found rapid uptake to sorption, with little or no subsequent ammonia loss. Their linear sorption $K_D = 0.083$ L/kg. (Sorption relationships are discussed in more detail in Chapter 10—the following discussion focuses in ammonia sorption only.)

$$\text{Gravel:} \quad 0.3-1.3 \text{ cm} \quad C_S = 0.083 C_L^{1.00} \tag{9.5}$$

Sikora *et al.* (1995b) provided data from which Freundlich constants could be fit:

$$\text{Fine gravel:} \quad 0.5-1.0 \text{ cm} \quad C_S = 0.77 C_L^{0.64} \tag{9.6}$$

$$\text{Coarse gravel:} \quad 0.5-2.0 \text{ cm} \quad C_S = 1.63 C_L^{0.55} \tag{9.7}$$

Weatherly and Miladinovic (2004) provided Langmuir constants for the zeolites clinoptilolite and mordenite:

$$\text{Clinoptilolite:} \quad \frac{1}{K} = 2.5 \text{ mg/L} \quad S_{\max} = 6.9 \text{ g/kg} \tag{9.8}$$

$$\text{Mordenite:} \quad \frac{1}{K} = 19.6 \text{ mg/L} \quad S_{\max} = 13.1 \text{ g/kg} \tag{9.9}$$

Lahav and Green (2000) provided Langmuir constants for the zeolite chabazite:

$$\text{Chabazite:} \quad \frac{1}{K} = 10.0 \text{ mg/L} \quad S_{\max} = 50.5 \text{ g/kg} \tag{9.10}$$

The median ammonia loading for HSSF systems is about 1.0 g/m^2·d, and the median concentration is 20 mg/L. For the parameters above, the equilibrium ammonia sorbed at 20 mg/L is 2–25 g/m^2 for a 60-cm deep bed. Therefore, the bed solids can hold approximately 2–25 days' supply of ammonia via sorption phenomena.

However, if the wetland substrate is exposed to oxygen, perhaps by periodic draining, sorbed ammonium may be oxidized to nitrate. Nitrate is not bound to the substrate, and is washed out by subsequent rewetting. This concept forms the basis for intermittently fed and drained, vertical flow treatment wetlands, and for other wetland systems that are alternately flooded and drained.

Ammonia Volatilization

Un-ionized ammonia is relatively volatile and can be removed from solution to the atmosphere through diffusion through water upward to the surface, and mass transfer from the water surface to the atmosphere.

Theoretical Considerations

Total dissolved ammonia exists in the two forms, free or un-ionized (NH_3), and ionized (NH_4^+). These interconvert readily in water, according to Equation 9.2, which allows the computation of the concentration of free ammonia in terms of total ammonia:

$$C_{AL} = \frac{C_{ATL}}{1+K_d} \tag{9.11}$$

where
C_{AL} = concentration of free ammonia in the bulk water, g/m³
C_{ATL} = concentration of total ammonia in the bulk water, g/m³

Free ammonia may also exist as a gas, whereas ionized ammonia is nonvolatile. The process of volatilization carries free ammonia from the water into the air above. That overall process comprises four major components in series (see Chapter 5): (1) partial conversion of ionized ammonia to free ammonia (dissociation), (2) diffusion of free ammonia to the air–water interface (water-side mass transfer), (3) release of free ammonia to the air at the interface (volatilization), and (4) diffusion of free ammonia from the air–water interface into the air above (air-side mass transfer). These component processes are conceptually well understood because of studies associated with ammonia stripping as an engineering technology.

The loss of free ammonia may be described by a two-film mass transfer equation (Welty *et al.*, 1983; Liang *et al.*, 2002):

$$J = k(C_{AL} - C_{AL}^{*}) \tag{9.12}$$

$$\frac{1}{K_L} = \frac{1}{k_L} + \frac{1}{Hk_G} \tag{9.13}$$

where
C_{AL}^{*} = water concentration of free ammonia that would be in equilibrium with the free ammonia in the bulk air, g/m³
H = Henry's Law coefficient, dimensionless
K_L = overall mass transfer coefficient, m/d
k_G = air-side mass transfer coefficient, m/d
k_L = water-side mass transfer coefficient, m/d

Water–air equilibrium, or solubility, is governed by Henry's law:

$$C_{AL}^{*} = \frac{C_{AG}}{H} \tag{9.14}$$

where
C_{AG} = concentration of free ammonia in the bulk air, g/m³

The value of H is temperature-dependent (Liang *et al.*, 2002):

$$H = \left(\frac{2.395\times10^5}{T+273.16}\right)\exp\left(\frac{-4151}{T+273.16}\right) \tag{9.15}$$

Under almost all circumstances, the ammonia concentration in the air above the wetland will be negligibly small, and hence may be presumed to be zero. Additionally, total ammonia rather than free ammonia is used in the overall vapor loss equation:

$$J = K_L C_{AL} \tag{9.16}$$

$$J = K_L\left(\frac{C_{ATL}}{1+K_d}\right) = kC_{ATL} \tag{9.17}$$

where
k = first-order volatilization rate constant based on total ammonia, m/d

There are two choices for a first-order removal rate: one based on the free ammonia concentration in the water (Equation 9.16), and one based on the total ammonia concentration in the water (Equation 9.17); the latter is used here.

Practical Application

Many factors influence component processes, most of which will not be known or measured for field situations involving treatment wetlands. Solubility depends on temperature, and degree of ionization depends on temperature and pH. However, the process of ammonia volatilization involves proton transfer, and a theoretical decrease in pH. Such a decrease has been observed in laboratory volatilization tests (Shilton, 1996). Additionally, both temperature and pH undergo large diurnal swings in some treatment wetlands up to 8°C and 2 pH units. In some few situations, there may be vertical stratification of the water column, leading to interfacial temperature and pH conditions that deviate from those in the bulk water (Jenter *et al.*, 2003).

The water-side mass transfer coefficient (k_L) depends upon the degree of turbulence (mixing) in the water, which in turn depends on depth, velocity, and the amount of submersed plant and litter material (Serra *et al.*, 2004), together with the wind speed (Liang *et al.*, 2002). The air-side mass

FIGURE 9.7 Ammonia losses were measured directly at ponds at Greensboro, North Carolina. (Photo courtesy M. Poach.)

transfer coefficient (k_G) depends upon the degree of turbulence (mixing) in the air, which in turn depends on wind speed and amount of emergent plant biomass. The studies of Liang *et al*. (2002) suggest that both air-side and water-side mass transfer resistance are important for ammonia losses from ponds. That is in contrast to the work of Freney *et al*. (1985), which suggested that for a rice crop, the mass transfer resistance was entirely in the air. Therefore, ammonia loss rates should depend not only upon temperature and pH, but also on site-specific conditions (see Figure 9.7).

Several studies of ammonia volatilization from ponds and wetlands provide data from which first-order rate constants may be calculated (Table 9.5). Values of k range from 0.11 to 28 m/yr, which is an unacceptably large range. A modified Arrhenius temperature factor developed from the data of Stratton (1969) is $\theta = 1.094$. This was used to adjust rate constants to 20°C in Table 9.5. The k_{20} values so computed for wetland systems span a much narrower range 0.28–0.68 m/yr, with mean ± SD = 0.47 ± 0.14. For pond systems, the values are much higher, mean ± SD = 4.2 ± 4.6. There is also a clear trend of increasing k with pH for ponds, which has been reported in several studies (Stratton, 1968; Shilton, 1996; Liang *et al*., 2002). The reduced rates for wetlands may be attributed to the vegetation, which breaks the wind and thus lowers both the water-side and air-side mass transfer coefficients. Presumably, there would be a pH effect for wetlands, but FWS wetland pH values are most often tightly clustered in the range 7.0–7.5, thus preventing the manifestation of a pH effect.

These considerations indicate that emergent FWS wetlands will lose much less ammonia to volatilization than will ponds. Therefore, inclusion of open water sections in FWS treatment wetlands encourages ammonia loss (Poach *et al*., 2004; see Figure 9.8). Volatilization rate constants for vegetated wetlands are quite small compared with rate constants for other mechanisms, as will be discussed in the following text. However, the same is not necessarily true for open water components.

Microbial Processes

Wetlands are a rich environment for a large suite of microbes that mediate or conduct numerous chemical reactions involving nitrogen. Heterotrophic bacteria derive carbon from preformed organic compounds, whereas autotrophs acquire energy and carbon from inorganic sources. Denitrification is often, but not always, accomplished by heterotrophs in wetlands, while nitrification is carried out autotrophically. Microbes also produce enzymes that can break down complex molecules, both inside and outside the cell. Microbes are preferentially associated with solid surfaces, rather than as free-floating organisms. The principal nitrogen microbial wetland processes are therefore carried out in biofilms located on soils, sediments, and submerged plant parts.

In the following sections, the principal nitrogen conversions are discussed in more detail (see Figure 9.5).

Ammonification of Organic Nitrogen

Ammonification is the biological transformation of organic nitrogen to ammonia and is the first step in mineralization of organic nitrogen (Reddy and Patrick, 1984). This process occurs both aerobically and anaerobically, and releases ammonia from dead and decaying cells and tissues. Heterotrophic microorganisms are considered to be the group involved (U.S. EPA, 1993b). The reactions can take place intracellularly or extracellularly, via the action of enzymes acting upon proteins, nucleic acids, and urea (Maier *et al*., 2000). The sources of nitrogenous organics are plant and animal tissues, and direct excretion of urea.

Typical ammonification reactions are:

Urea breakdown

$$NH_2CONH_2 + H_2O \rightarrow 2NH_3 + CO_2 \tag{9.18}$$

Amino acid breakdown

$$RCH(NH_2)COOH + H_2O \rightarrow NH_3 + CO_2 \tag{9.19}$$

TABLE 9.5
Rate Constants for Ammonia Volatilization for Ponds and Wetlands

Site	T (°C)	pH	Total NH_3–N (g/m^3)	Un-ionized NH_3–N (g/m^3)	Loss rate ($g/m^2 \cdot yr$)	Total NH_3–N k (m/yr)	Total NH_3–N k_{20} (m/yr)	Reference
Duplin County, North Carolina								Poach *et al*. (2002, 2003) Field: large-scale chambers
Grass	23	7.2	55	0.51	30	0.46	0.35	
Bulrush	24	7.2	46	0.42	19	0.40	0.28	
Cattail	22	7.2	46	0.36	20	0.57	0.48	
Greensboro, North Carolina								Poach *et al*. (2004) Field: large-scale chambers
Bulrush and Cattail	23	7.0	60	0.54	46	0.70	0.53	
Pond	25	7.4	40	1.08	237	10.4	6.6	
Ujjain, India								Billore *et al*. (1994) Field: small-scale chambers
Water	35	—	4	—	15	4.2	1.09	
Duckweed	35	—	7	—	46	6.5	1.69	
Cattail	35	—	14	—	37	2.6	0.68	
New Zealand								Shilton (1996) Lab: small-scale chambers
Pond	20	8.6	549	86	389	0.69	0.69	
Al-Bireh, Palestine								Zimmo *et al*. (2003) Field: small-scale chambers
Duckweed	17	7.8	50	1.12	5.4	0.11	0.14	
Pond	17	8.1	38	1.38	6.7	0.19	0.25	
Griffith, Australia								Freney *et al*. (1985) Field: air-side measurements
Rice	20	8.0	73	2.78	34.9	0.48	0.48	
San Diego, California								Stratton (1969) Lab: flow chambers
Pond	29	9.8	0.47	0.39	0.89	28	12.5	
Pond	32	9.1	1.75	0.92	37	20	6.8	

Note: Values based on total ammonia are shown.

It is curious that the wastewater treatment literature does not directly address ammonification, despite the considerable proportion of organic nitrogen in raw wastewaters. The ammonification step is identified on diagrams, but no mention of chemistry or rates is found in manuals (Brown and Caldwell, 1975; U.S. EPA, 1993b) or texts (Metcalf and Eddy Inc., 1991). In some instances, it is recommended to lump organic and ammonium (as TKN) in calculations of

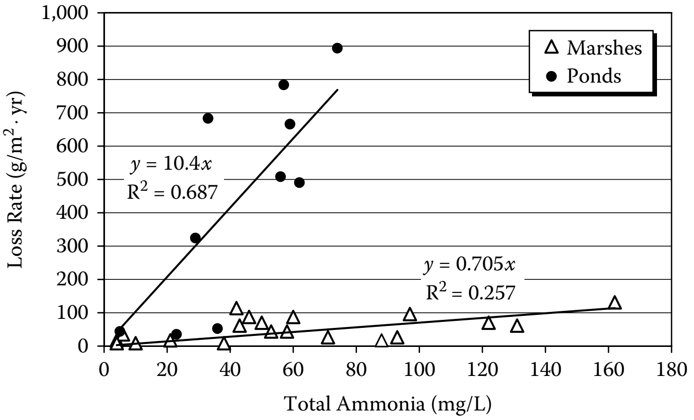


FIGURE 9.8 Ammonia volatilization losses from 12 marshes and 6 ponds at Greensboro, North Carolina. Conditions in the marshes were $T = 23°C$, pH = 7.0; in the ponds $T = 25°C$, pH = 7.4; wind was 0.2–1.5 m/s. (Replotted from Poach *et al*. (2003) *Ecological Engineering*, 20(2): 183–197, with zero intercept.)

ammonia processing, on the presumption that organic nitrogen will add to the potential ammonia concentrations (U.S. EPA, 2000a). That procedure can be misleading for two reasons. First, ammonification is not instantaneous, and conversion proceeds at rates that influence the removal of TKN in many instances. Kinetically, ammonification proceeds more rapidly than nitrification, thus creating the potential for increasing ammonia concentrations along the flow-path of a wetland and requiring design for nitrogen removal to include both ammonification and the slower nitrification process. Second, the ammonification process does not proceed to completion in wetlands, although the removal of ammonia can go to completion for long enough detention. There is an organic nitrogen background concentration which may consist of irreducible residuals, or be due to return fluxes of organic nitrogen from decomposing solids.

Nitrification of Ammonia

Nitrification is the principal transformation mechanism that reduces the concentration of ammonia nitrogen in many wetland treatment systems, by converting ammonia nitrogen to oxidized nitrogen, van de Graaf *et al.* (1996) defined nitrification as the biological formation of nitrate or nitrite from compounds containing reduced nitrogen with oxygen as the terminal electron acceptor. Nitrification has been typically associated with the chemoautotrophic bacteria, although it is now recognized that heterotrophic nitrification occurs and can be of significance (Keeney, 1973; Paul and Clark, 1996).

Results from Conventional Wastewater Treatment Processes

Biological nutrient removal systems may be broadly categorized as suspended growth (e.g., activated sludge) or attached growth (e.g., trickling filters). In such devices, nitrification is considered to be a two-step, microbially mediated process in U.S. EPA (1993b):

$$\text{Nitritation}\quad 2NH_4^+ + 3O_2 \xrightarrow{\textit{Nitrosomonas}} 2NO_2^- + 2H_2O + 4H^+ \tag{9.20}$$

$$\text{Nitrification}\quad 2NO_2^- + O_2 \xrightarrow{\textit{Nitrobacter}} 2NO_3^- \tag{9.21}$$

The first step, nitritation, is mediated primarily by autotrophic bacteria in the genus *Nitrosomonas* and the second step, nitrification, by bacteria in the genus *Nitrobacter*. Both steps can proceed only if oxygen is present, and thus the actual nitrification rate may be controlled by the flux of dissolved oxygen into the system.

Based on this stoichiometric relationship, the theoretical oxygen consumption by the first nitritation reaction is about 3.43 g O_2 per gram of NH_3–N oxidized, and 1.14 by the second nitrification reaction, for a total of 4.57. Actual consumption is reportedly somewhat less, 4.3 g O_2 per gram of NH_3–N oxidized (Metcalf and Eddy Inc., 1991). The oxidation reactions release energy used by both *Nitrosomonas* and *Nitrobacter* for cell synthesis. The combined processes of cell synthesis create 0.17 g of dry weight biomass per gram of ammonia nitrogen consumed (U.S. EPA, 1993b). Nitrification of ammonia to nitrate consumes approximately 7.1 g of alkalinity (as $CaCO_3$) for each nitrified gram of ammonia nitrogen, as two moles of H^+ are released for each mole of ammonia nitrogen consumed in Equation 9.20 (U.S. EPA, 1993b). Thus nitrification lowers the alkalinity and pH of the water.

The optimal pH range observed for nitrification in suspended growth treatment systems is between about 7.2 and 9.0 (Metcalf and Eddy, Inc., 1991). Treatment wetlands almost always operate at circumneutral pH (see Chapter 5); consequently, this factor should be a minor influence on nitrification in those systems.

Wetland Environments

Natural environments are considerably more complex than the situations in biological nutrient removal systems in conventional wastewater treatment plants (WWTPs). There are now enough wetland data to begin to understand some differences, and to appreciate that WWTP results may not apply to wetlands.

There are more genera potentially involved in natural systems than those identified above. Ammonia oxidizing bacteria (AOB) include *Nitrosospira* and *Nitrosococcus* in addition to *Nitrosomonas* (Bothe *et al.*, 2000). Austin *et al.* (2003) found *Nitrosospira* just as abundant as *Nitrosomonas* in a treatment wetland, with lesser numbers of *Nitrosococcus*. Likewise, nitrite is oxidized by *Nitrospira* as well as *Nitrobacter*, and the former was found to be much more prevalent in a treatment wetland (Austin *et al.*, 2003). Furthermore, heterotrophic bacteria are capable of nitrification, such as *Paracoccus denitrificans* and *Pseudomonas putida* (Bothe *et al.*, 2000). Nevertheless, *Nitrosomonas* is found in treatment wetlands (Silyn-Roberts and Lewis, 2001). The oxidation of ammonia to nitrite in natural systems is suggested to comprise two steps, not one (Bothe *et al.*, 2000), catalyzed by enzymes:

$$NH_3 + O_2 + 2H^+ + 2e^- \xrightarrow[\text{monooxygenase}]{\text{Ammonia}} NH_2OH + H_2O \tag{9.22}$$

$$NH_2OH + H_2O \xrightarrow[\text{oxidoreductase}]{\text{Hydroxylamine}} NO_2^- + 5H^+ + 4e^- \tag{9.23}$$

This scheme suggests that hydroxylamine is an intermediate in the process, which presents alternate nitrogen processing possibilities. Further, one of the oxygen atoms in nitrite derives from O_2, the other from water.

Nitrite oxidizing bacteria (NOB) were found not to include *Nitrobacter* in two FWS treatment wetlands (Flood *et al.*, 1999). Similarly, Austin *et al.* (2003) found *Nitrospira* (4% of total) to be much more abundant than *Nitrobacter*

(0.1% of total) in a treatment wetland. Importantly, nitrite may be also be destroyed by processes other than conversion to nitrate, as shall be discussed in a later section.

On a practical level, these considerations cast doubt about the applicability to wetlands of the stoichiometry advocated for WWTP environments (Equations 9.20 and 9.21). For instance, the dissolved oxygen requirement for Equations 9.22 and 9.23 is 1.14 g O_2 per gram of ammonia nitrogen, rather than the 3.43 suggested by Equation 9.20. Alkalinity requirements are also greatly reduced. The stoichiometric factor of 4.3 g O_2 per gram of NH_4–N oxidized has been used in many treatment wetland publications as a means of inferring the maximum amount of oxygen transferred into the water (e.g., Platzer, 1999; Cooper, 2001, 2005). But, in many wetland situations, the 4.3 factor does not seem to be applicable (Tanner and Kadlec, 2002). These alternative pathways with the potential to substantially reduce the oxygen fluxes required to drive NH_4–N removal need to be investigated further in both natural and constructed wetlands to develop an understanding of their role in wetland nitrogen removal.

The necessity of a low carbon-to-nitrogen ratio, another concept from activated sludge and attached growth technologies, appears dubious for wetlands. It has been suggested that the biochemical oxygen demand (BOD) level "must be below (BOD/TKN < 1.0)" for "successful nitrification" in treatment wetlands (Reed *et al*., 1995; Crites *et al*., 2006). In conventional devices, the carbon consumption activity of heterotrophs may cause them to dominate the overall bacterial population, but with a smooth transition from 3% to 35% nitrifiers as the BOD_5:TKN ratio decreases from 9 to 0.5 in activated sludge plants (Metcalf and Eddy Inc., 1991). Similarly, the result is a smooth decrease in nitrification rates in attached growth systems, from a relative level of 100% in the absence of BOD to 40% at BOD_5:TKN = 5.0 (Brown and Caldwell, 1975).

Free water surface treatment wetlands operate with a variety of inlet carbon-to-nitrogen ratios, ranging from 0.28 to 4.41 (5th to 95th percentiles, N = 126 wetlands). The mean inlet ratio is 2.0, and the mean outlet ratio is 1.6. Only one third of the 126 FWS wetlands met the criterion BOD:TKN < 1.0. This distribution is rather narrow, and would not lead to marked differences in potential nitrification rates. Considering direct evidence, there is essentially no correlation between the BOD:TKN ratio and measures of nitrification performance. For example, the TKN load removed versus BOD:TKN ratio has an $R^2 = 0.037$. Transect data sets display no nitrogen removal lag as carbon is removed (Tanner *et al*., 2002a). Therefore, it is not reasonable to accept this ratio as a controlling factor in FWS wetlands.

Denitrification

Denitrification is most commonly defined as the process in which nitrate is converted into dinitrogen via intermediates nitrite, nitric oxide, and nitrous oxide (Hauck, 1984; Paul and Clark, 1996; Jetten *et al*., 1997).

Denitrification (nitrate dissimilation) is carried out by facultative heterotrophs, organisms that can use either oxygen or nitrate as terminal electron acceptors. Starting from nitrate via nitrite, there is sequential production of nitric oxide (NO), nitrous oxide (N_2O), and nitrogen gas (N_2) (e.g., Cox and Payne, 1973; Koike and Hattori, 1978):

$$2NO_3^- \rightarrow 2NO_2^- \rightarrow 2NO \rightarrow N_2O \rightarrow N_2 \qquad (9.24)$$

Diverse organisms are capable of denitrification. In an array are organotrophs (e.g., *Pseudomonas, Alcaligenes, Bacillus, Agrobacterium, Flavobacterium, Propionibacterium, Vibrio*), chemolithotrophs (e.g., *Thiobacillus, Thiomicrospira, Nitrosomonas*), photolithotrophs (e.g., *Rhodopseudomonas*), diazotrophs (e.g., *Rhizobium*, *Azospirillum*), archaea (e.g., *Halobacterium*), and others such as *Paracoccus* or *Neisseria* (Focht and Verstraete, 1977; Knowles, 1982; Killham, 1994; Paul and Clark, 1996).

Results from Conventional Wastewater Treatment Processes

The overall stoichiometric nitrate dissimilation reaction based on methanol (CH_3OH) as a carbon source is summarized by the following (U.S. EPA, 1993b):

$$\begin{aligned} NO_3^- + 0.833\,CH_3OH \rightarrow\ & 0.5\,N_2 + 0.833\,CO_2 \\ & + 1.167\,H_2O + OH^- \end{aligned} \qquad (9.25)$$

Other carbon sources also may drive denitrification, such as glucose (Reddy and Patrick, 1984):

$$\begin{aligned} NO_3^- + 0.208\,C_6H_{12}O_6 \rightarrow\ & 0.5\,N_2 + 1.25\,CO_2 \\ & + 0.75\,H_2O + OH^- \end{aligned} \qquad (9.26)$$

The carbon (energy) requirements are 1.90 g methanol and 2.67 g glucose per gram of nitrate nitrogen, respectively. Some nitrate and carbon are also used by denitrifying bacteria for cell synthesis. For instance, another 0.57 g methanol is required for bacterial growth, bringing the total to 2.47 g of methanol to support the denitrification of 1 g of nitrate nitrogen. This translates to an optimum carbon level of 2.3 g BOD per g NO_3–N (Gersberg *et al*., 1984). In the absence of this or another equivalent carbon source, denitrification is inhibited.

As indicated by Equations 9.25 and 9.26, denitrification produces alkalinity. The observed yield of this process is about 3.0 g alkalinity as $CaCO_3$ per gram of NO_3-N reduced. This increase in alkalinity is accompanied by an increase in the pH of the wetland surface water.

Theoretically, denitrification does not occur in the presence of dissolved oxygen. However, denitrification has been observed in suspended and attached growth treatment systems that have relatively low measured dissolved oxygen concentrations, but not above 0.3–1.5 mg/L (U.S. EPA, 1993b).

This is presumably due in part to the activity of aerobic denitrifiers, such as *Paracoccus denitrificans*.

Wetland Environments—Carbon Sources

The carbon source in wetlands is neither methanol nor glucose, but rather organic matter that is sometimes characterized by the Redfield ratio C:N:P = 106:16:1 (Davidsson and Stahl, 2000). The denitrification reaction is then written:

$$84.8\,NO_3^- + (CH_2O)_{106}(NH_3)_{16}(H_3PO_4) \rightarrow$$
$$42.4\,N_2 + 106\,CO_2 + 16\,NH_3 + 148.8\,H_2O + H_3PO_4 \tag{9.27}$$

This reaction is irreversible in nature, and occurs in the presence of available organic substrate only under anaerobic or anoxic conditions (E_h = +350 to +100 mV), where nitrogen is used as an electron acceptor in place of oxygen. More and more evidence is being provided from pure culture studies that nitrate reduction can occur in the presence of oxygen. Hence, in waterlogged soils, nitrate reduction may also start before the oxygen is depleted (Kuenen and Robertson, 1987; Laanbroek, 1990).

The carbon (energy) requirement is 3.02 g organic matter per gram of nitrate nitrogen. Further, some ammonia is theoretically liberated, which can support growth or add to the overall wetland ammonia pool.

As most denitrification is accomplished by heterotrophic bacteria, the process is strongly dependent on carbon availability. There is a general correlation between total soluble organic matter content and denitrification potential, but much better correlation occurs with the supply of easily decomposable organic matter or water-extractable organic carbon (Bremner and Shaw, 1958; Broadbent and Clark, 1965; Paul and Clark, 1996). Organic substances able to act as sources of energy and as hydrogen donors may be present in sediments and soils through the decomposition of tissues or be provided by living roots exudates (Stefanson, 1973; Bailey, 1976).

A number of treatment wetland studies have investigated the use of carbon supplements in the form of added plant biomass (Gersberg *et al*., 1983, 1984; Burchell *et al*., 2002; Hume *et al*., 2002a). Another study added methanol (Gersberg *et al*., 1983), with good effect. Burgoon (2001) provided carbon by feed-forward of un-nitrified influent to wetlands receiving nitrified potato processing waters. All such studies have shown that carbon can be limiting in wetlands at high nitrate loadings. The amount of total carbon in dead and decomposing biomass is on the order of 40% of the dry biomass (Ingersoll and Baker, 1998; Baker, 1998; Hume *et al*., 2002b). Not all of the total carbon produced is available for denitrifiers. Baker (1998) has suggested that the C:N loading ratio be at least 5:1 so that carbon does not become limiting, which in his work translated to 20% availability. Hume *et al*. (2002b) suggest 8% availability. Presuming a carbon content of 40%, the required productivities are at the lower end of the range for emergent marshes (Kadlec and Knight, 1996). However, realization of higher nitrate removal rates, corresponding to higher inlet concentrations, may stress the ability of the wetland to generate the required carbon energy source. If carbon is limiting, the rate of denitrification will depend strongly on the rate of carbon supply (Hume *et al*., 2002a).

It should be noted that the most labile form of organic carbon in wetland environments is the influent BOD, which is likely used preferentially (when available) to reduce oxidized forms of nitrogen.

Wetland Environments—Oxygen Inhibition

Denitrification has been observed in numerous wetland treatment systems which have considerable dissolved oxygen in their surface waters (Van Oostrom and Russell, 1994; Phipps and Crumpton, 1994). This apparent anomaly is due to the complicated spatial zonation in a wetland. Oxygen gradients occur between surface waters and bottom sediments in wetlands, allowing both aerobic and anoxic reactions to proceed in close vertical proximity (millimeters) near the sediment–water interface (Figure 9.9). Thus, nitrate formed by nitrification in surface waters may diffuse into top anoxic soil layers where it is effectively denitrified (Reddy and Patrick, 1984).

Significant quantities of oxygen pass down through the airways to the roots (Brix and Schierup, 1990; Brix, 1993); and significant quantities of other gases, such as carbon dioxide and methane, pass upward from the root zone. Some—perhaps most—of the oxygen passing down the plant into the root zone is used in plant respiration (Brix, 1990). However, there is a great deal of chemical action in the microzones near the roots of wetland plants. Figure 9.10 shows that the oxygenated microzone around a rootlet can conduct nitrification reactions, whereas denitrification reactions can be occurring only microns away in the anaerobic bulk soil. Diffusion easily connects these zones because of their close proximity.

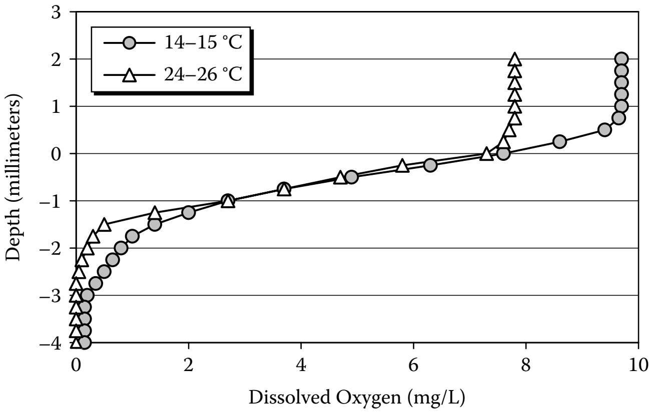


FIGURE 9.9 Oxygen distribution above and below the sediment–water interface at two different temperatures. (Data from Crumpton and Phipps (1992) *The Des Plaines River Wetlands Demonstration Projects*. Vol. III, chap 5. Wetlands Research, Inc., Chicago, Illinois.)

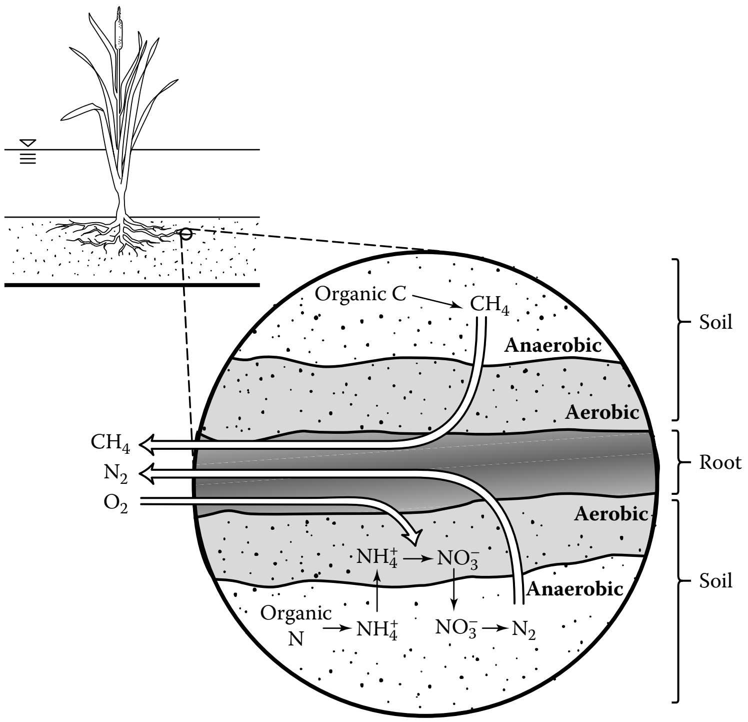


FIGURE 9.10 Pathways of nitrogen transformations in the immediate vicinity of a plant root.

Bacteria attached to surfaces are usually more numerous than free-living (planktonic) bacteria (Bastviken *et al.*, 2003, 2005). Attached bacteria form microbial communities that are embedded in polysaccharide matrixes, e.g., biofilms, and the bacterial activity within these biofilms is regulated by diffusion of nutrients into the biofilm and by internal processes within this layer. In wetlands, these surfaces are as important as the sediment for the nitrogen turnover processes (Eriksson and Weisner, 1997; Eriksson, 2001). Biofilms, therefore, comprise a third type of spatial nonuniformity in the wetland environment. Diffusion within the biofilm controls the internal supplies of oxygen, nitrate, and ammonia, thus regulating the net effects of bacterial conversions. In surface flow treatment wetlands, biofilms have been found to contain 10^8–10^9 organisms/cm^2, mostly beta and gamma *Proteobacteria* (Flood *et al.*, 1999). Ammonia oxidizers (beta) were more prevalent near the inlet; denitrifiers (gamma) were more prevalent near the outlet. Alum addition was found to totally eliminate these bacteria.

Another type of spatial nonuniformity exists due to the presence of longitudinal gradients in dissolved oxygen in the flow direction. Oxygen may be depleted by heterotrophic activity, as well as nitrification; but atmospheric reaeration also occurs.

Clearly, wetland oxygen environments are much more complex than either the complete-mix situation that dominates activated sludge processing or the attached growth environment of trickling filters. Results from those technologies should not be extrapolated to treatment wetlands.

Wetland Environments—Dissimilatory Nitrate Reduction to Ammonium Nitrogen

Nitrate loss in treatment wetlands is often attributed to denitrification in the absence of proof that this mechanism is indeed the operative one. Other known and studied candidate mechanisms in wetlands include assimilation by plants and microbiota, and dissimilatory reduction to ammonium nitrogen (DNRA). These alternative reduction routes have been documented to comprise from 1–34% of the total nitrate loss (Bartlett *et al.*, 1979; Stengel *et al.*, 1987; Cooke, 1994; Van Oostrom and Russell, 1994). Bartlett *et al.* (1979) measured production of ammonium, dinitrogen, and nitrous oxide for microcosms with soils from a treatment wetland, but with no plants. From 1–6% of the product was ammonium nitrogen; the balance was measured as dinitrogen, with only trace amounts of nitrous oxide. Cooke (1994) measured ^{15}N-labelled nitrate, ammonium, and organic nitrogen in unvegetated microcosms in a treatment wetland. He found 34%, 6%, and 60% of $K^{15}NO_3$ converted by dissimilatory processes, microbial assimilatory processes, and denitrification, respectively, at one site; and 25%, 5%, and 70% at a second site. Stengel *et al.* (1987) used the acetylene blockage technique to establish that 75–90% of the nitrate loss in a flow through, *Phragmites*/gravel SSF unit was due to denitrification. Van Oostrom and Russel (1994) measured 16% dissimilatory nitrate reduction in microcosms containing *Glyceria maxima* mats.

The relative importance of denitrification and dissimilatory reduction of nitrate to ammonium in the soil environment is far from certain. Denitrification may be the dominant process in environments rich in nitrate but poor in carbon, whereas the dissimilatory reduction of nitrate and nitrite to ammonium tends to dominate in carbon-rich environments, which are preferably colonized by fermentative bacteria (Tiedje *et al.*, 1982). So nitrate-ammonifying bacteria may be favored by nitrate-limited conditions (Laanbroek, 1990). Nitrate ammonification is found in facultative anaerobic bacteria belonging to the genera *Bacillus, Citrobacter,* and *Aeromonas*, or in the members of Enterobacteriaceae (Cole and Brown, 1980; MacFarlane and Herbert, 1982; Grant and Long, 1985). However, strictly anaerobic bacteria belonging to the genus *Clostridium* are also able to reduce nitrate to ammonia (Caskey and Tiedje, 1979, 1980). For many of the bacteria responsible for dissimilation to ammonium, formate is a major electron donor both for nitrate and nitrite, although most of the research on the nitrate reductase activity has been restricted to enteric bacteria such as *Escherichia coli* (Killham, 1994).

Conversion of NO_3^- to NH_4^+ and organic nitrogen increases markedly with decreasing redox potential, high pH, and large quantities of readily oxidizable organic matter (Nommik, 1956; Buresh and Patrick, 1978, 1981). Nitrate respiration to NH_4^+ occurs at E_h values of less than −100 mV (Patrick, 1960; Buresh and Patrick, 1981).

Wetland Environments—Effects of Vegetation

Wetland vegetation influences nitrogen supplies because of uptake associated with growth, which is the topic of a later section. However, vegetation also serves other functions in nitrate reduction, including carbon supply and microbial attachment sites. Wetlands may contain emergent or submergent vegetation, and areas of unvegetated open water. Plants may be woody or soft-tissued. Community specificity for denitrification is expected, roughly correlated with carbon availability and the amount of immersed surface area.

Unvegetated open water does not promote denitrification, resulting in rate constants about one third of those for vegetated systems (Arheimer and Wittgren, 1994). Smith *et al.* (2000) have shown nitrate removal proportional to number of shoots in a *Schoenoplectus* spp. wetland. Wetlands with woody species—shrubs and trees—also have relatively low rates of denitrification (Westermann and Ahring, 1987; DeLaune *et al.*, 1996). Carbon limitation is the likely cause.

Either emergent or submergent vegetation can harbor epiphytic microbial biofilms on living and dead plant material (Eriksson and Weisner, 1997). However, living underwater plants produce oxygen, which inhibits denitrification. Field data do not provide clear guidance on the choice between emergent and submergent plants. Weisner *et al.* (1994) found *Potamageton* to be more effective than *Glyceria*, and *Phragmites* stands to be better than open water. Eriksson and Weisner (1997) measured very high rates of denitrification in a reservoir with dense *Potamageton pectinatus*. Conversely, Gumbricht (1993a) found low rates for *Elodea canadensis*. Toet (2003) found that emergent stands of *Typha* and *Phragmites* yielded nitrate removal rates of 98 and 287 kg/ha·yr, respectively, whereas mixed submerged aquatics (*Elodea, Potamogeton* and *Ceratophyllum*) removed only 16–20 kg/ha·yr.

These considerations lead to the conclusion that fully vegetated marshes with either emergent or submergent communities are the preferred option for denitrification. Weisner *et al.* (1994) reached this conclusion and suggested that an alternating banded pattern perpendicular to flow would additionally provide hydraulic benefits.

Denitrifying bacteria are more abundant than the nitrifiers, in both FWS and SSF treatment wetlands. Listowel results show higher populations in the sediments in spring and summer, about 10^6/g versus 10^5/g in fall and winter (Herskowitz, 1986). Denitrifiers were found at higher levels in a U.K. gravel bed, approximately 10^7–10^8/g; and most were associated with roots rather than the gravel (May *et al.*, 1990).

Sulfur-Driven Autotrophic Denitrification

Sulfur-driven autotrophic denitrification, as an alternate to carbon-driven, heterotrophic denitrification, is well known (Koenig and Liu, 2001; Soares, 2002). The bacterium *Thiobacillus denitrificans* can reduce nitrate to nitrogen gas while oxidizing elemental sulfur, or reduced sulfur compounds including sulfide (S^{2-}), thiosulfate ($S_2O_3^{2-}$), and sulfite (SO_3^{2-}). For example, the chemistry proposed for utilization of elemental sulfur is (Batchelor and Lawrence, 1978):

$$\begin{gathered} NO_3^- + 1.1\,S + 0.40\,CO_2 + 0.76\,H_2O + 0.08\,NH_4^+ \rightarrow \\ 0.5\,N_2 + 0.08\,C_5H_7O_2N + 1.1\,SO_4^{2-} + 1.2\,H^+ \end{gathered} \tag{9.28}$$

If sulfide is the primary species of reduced sulfur, the proposed chemistry is (Komor and Fox, 2002):

$$\begin{gathered} NO_3^- + 0.74\,S^{2-} + 0.1886\,CO_2 \rightarrow 0.48\,N_2 + 0.74\,SO_4^{2-} \\ + 0.037\,C_5H_7O_2N + 0.1\,H^+ + 0.37\,H_2O \end{gathered} \tag{9.29}$$

This reaction requires 1.69 g sulfide sulfur per gram of nitrate nitrogen. Other postulated reactions also exist. For instance, iron pyrite may be oxidized (Pauwels and Talbo, 2004):

$$14\,NO_3^- + 5\,FeS_2 + 4\,H^+ \rightarrow 7\,N_2 + 5\,Fe^{2+} + 10\,SO_4^{2-} + 2\,H_2O \tag{9.30}$$

Treatment wetlands can have many forms of sulfur in sediments, arising from the introduction of sulfate in the incoming water. Reducing conditions can form sulfides and elemental sulfur in the sediments (see Chapter 11). Those sediments also contain carbon compounds, and consequently both heterotrophic, carbon-driven, and sulfur-driven denitrification have been observed to occur simultaneously in wetland sediments (Nahar *et al.*, 2000; Komor and Fox, 2001, 2002; Wass, 2003). The production of dinitrogen gas is accompanied by oxidation of sulfide to sulfate by the autotrophic process.

Given the variety of alternate electron acceptors for denitrifying organisms, it is not surprising that carbon is not limiting in some wetland situations where it would be expected (Fleming-Singer and Horne, 2006).

Aerobic Denitrification

Nitrite reduction to gaseous products by denitrifying bacteria used to be considered to be a strictly anaerobic process, but this fallacy was dispelled with the discovery of aerobic denitrification (Robertson *et al*., 1995). Aerobic denitrification is often coupled to heterotrophic nitrification in one organism. Because nitrification is mostly measured by the formation of nitrate or nitrite under oxic conditions, although (aerobic) denitrification is not expected under such conditions, this coupled process is not easily observed in standard enrichment cultures. The observation that *Thiosphaera pantotropha* and other organisms are not only heterotrophic nitrifiers, but also aerobic denitrifiers forced a reevaluation of this approach (Ludwig *et al*., 1993; Jetten, 2001). Aerobic denitrifiers are present in high number in natural soil samples. Even though the specific activities are not always very high, they are sufficient to allow significant contribution to the turnover of compounds in the nitrogen cycle (Jetten *et al*., 1997).

Anaerobic Ammonia Oxidation (Anammox)

There is now solid evidence for anaerobic elimination of nitrite by ammonia, also called *anaerobic ammonia oxidation* (anammox), in a number of wastewater treatment environments (van de Graaf *et al*., 1990; Mulder *et al*., 1995; van Loosdrecht and Jetten, 1998). In an environment with nitrite and ammonia present, a reaction to dinitrogen has been demonstrated commercially:

$$NH_4^+ + NO_2^- \xrightarrow[\textit{Nitrosomonas eutropha}]{\textit{Planctomycetes}} N_2 + 2H_2O \qquad (9.31)$$

The overall chemistry, including nitrite formation and bacterial growth requirements, has been proposed to be (Furukawa *et al*., 2001):

$$NH_3 + 0.85\ O_2 \rightarrow 0.44\ N_2 + 0.11\ NO_3^- + 1.43\ H_2O + 0.14\ H^+ \qquad (9.32)$$

The process proceeds through nitrite, formed according to Equations 9.22 and 9.23, and carries an oxygen requirement of only 1.94 g O per gram of NH_4–N. It is autotrophic, and has no organic carbon requirement.

Various commercial processes are now available which capitalize on the advantages of this alternative route for nitrogen removal. The completely autotrophic nitrogen removal over nitrite (CANON) process utilizes partial nitritation accompanied by Anammox® in a single vessel (Third *et al*., 2005). The SHARON® Anammox process utilizes partial nitritation in one vessel, and anaerobic elimination of nitrite by ammonia in a second (van Dongen *et al*., 2001). The microbiology has also been demonstrated in sequencing batch reactors (Kuai and Verstraete, 1998; Strous *et al*., 1998; Sliekers *et al*., 2002), activated sludge (Hao and van Loosdrecht, 2004), and rotating biological contactors (RBCs) (Helmer and Kunst, 1998; Koch *et al*., 2000).

Given advances in the ability to search for and detect nitrogen processing organisms, they have also been found in natural treatment systems. Anammox bacteria are present in soil aquifer treatment (Fox and Gable, 2003; Gable and Fox, 2003). They have also been identified in both FWS and SSF wetlands. Austin *et al*. (2003) found 13% of *Plantomycetes* in a vertical flow SSF wetland, of which a small fraction were autotrophic denitrifiers. They were also found in SSF and FWS wetlands treating partially nitrified domestic wastewater (Shipin *et al*., 2004).

The importance of this alternative pathway for ammonia and oxidized nitrogen removal for treatment wetland analysis lies in the reduced carbon and oxygen requirements: less than half the oxygen and no carbon, compared to conventional routes. In many wetland situations, there is adequate oxygen present to allow traditional nitrification (Equations 9.20 and 9.21). Likewise, in other instances, there is adequate carbon present to fuel traditional denitrification (Equation 9.27). But there are wetlands for which ammonia and oxidized nitrogen are removed in amounts that considerably exceed the estimated supplies of carbon and oxygen. Tanner and Kadlec (2002) found ammonia losses that would have required far more oxygen transfer than could reasonably be expected in a VF (saturated upflow) system, and Sun and Austin (2006), demonstrated similar results for highly loaded VF (saturated downflow) columns, while Bishay and Kadlec (2005) found the same for an FWS wetland. In the latter case, nitrite was present in relatively large quantities, and the carbon supply was not adequate to support traditional denitrification. In these instances, Anammox offers a potential explanation, but has not been confirmed.

Nitrogen Fixation

Biological nitrogen fixation is the process by which nitrogen gas in the atmosphere diffuses into solution and is reduced to ammonia nitrogen by autotrophic and heterotrophic bacteria, cyanobacteria (blue-green algae), and higher plants. The reduction of gaseous nitrogen (N_2) to ammonia (NH_3) takes place very rapidly and for this reason the individual steps in the reaction have not been investigated in detail. It is supposed that the whole reaction is a three-step, two-electrons-per-step mechanism (Winter and Burris, 1976):

$$N \equiv N \rightarrow \underset{\textit{diimide}}{HN = NH} \rightarrow \underset{\textit{hydrazine}}{H_2N - N_2H} \rightarrow NH_3 \qquad (9.33)$$

There are six main types of N_2-fixing organisms that can be found in soil (Killham, 1994):

1. Free-living bacteria such as *Bacillus*, *Klebsiella,* and *Clostridium* that fix N_2 anaerobically (the first two are facultative anaerobes and fix nitrogen under reduced oxygen tensions whereas *Clostridium* is an obligate anaerobe)
2. Bacteria of the genus *Rhizobium*, which fix N_2 mainly in the root nodules of leguminous plants
3. Actinomycetes of the genus *Frankia*, which fix N_2 in the root nodules of nonleguminous angiosperms such as *Alnus glutinosa* (those associations are often referred to as "actinorhizas")
4. Free-living cyanobacteria on the soil surface such as *Nostoc* and *Anabaena*
5. Symbiotic cyanobacteria found in the lichen symbiosis
6. N_2-fixing bacteria loosely associated with the roots of certain plants, sometimes referred to as "rhizocoenoses" (e.g., *Azotobacter*, *Beijerinckia* and *Azospirilllum*)

In wetland systems, free-living bacteria, cyanobacteria (blue-green algae), N_2-fixing bacteria loosely associated with the roots of certain plants, and probably *Frankia* are the most important N_2-fixing organisms.

Also, the aquatic fern, *Azolla*, and a few transitional, wetland vascular plant species in the genera *Alnus* and *Myrica* have been observed to fix atmospheric nitrogen (Waughman and Bellamy, 1980). Because nitrogen fixation uses stored energy from either autotrophic or heterotrophic sources, it is not an adaptive process when nitrogen is otherwise available for growth. The presence of ammonium nitrogen is reported to inhibit nitrogen fixation (Postgate, 1978; as referenced by Van Oostrom and Russell, 1994).

Under anaerobic conditions, microbial assemblages in the root zone of *Typha* spp. and *Glyceria borealis* were shown to fix considerable quantities of atmospheric nitrogen (Bristow, 1974). The majority of the activity was shown to be associated with the plants rather than the soils. Fixation rates at 20°C were determined to be 33.6 and 353 mg/kg roots·day for *Typha* and *Glyceria*, respectively. The measured rates of nitrogen fixation were estimated to be able to supply 10–20% of the growth requirement for *Typha*, and 100% for *Glyceria*. Under aerobic conditions, fixation dropped by an order of magnitude.

The nitrogen fixation potential for the soil-microbe assemblage was studied for 45 sites in 17 peatlands in eight countries by Waughman and Bellamy (1980). The appropriate subset in the context of treatment wetlands was the rich or extremely rich fen category, with $6.5 \leq \text{pH} \leq 7.6$, for which N = 12 sites. These showed fixation potentials averaging 0.622 mg/L per day of soil. A 30-cm root zone would then fix 70 gN/m²·yr. Other estimates from natural freshwater wetlands range from 0 to 55 gN/m²·yr (Vymazal, 2001b). Estimates of nitrogen fixation in a cypress dome receiving municipal wastewater ranged from 0.012 to 0.19 g/m²·yr (Dierberg and Brezonik, 1984) and were concluded to be an insignificant component of the TN loading to this treatment wetland.

These results do not permit quantification of the fixation occurring in treatment wetlands, but do indicate the ability of wetland plants and soils to fix nitrogen. It is unlikely that the rates of fixation in treatment wetlands contribute materially to nitrogen cycling in nitrogen-rich systems.

9.4 VEGETATION EFFECTS ON NITROGEN PROCESSING

Plants utilize nitrate and ammonium, and decomposition processes release nitrogen back to the water. There are two direct effects of vegetation on nitrogen processing and removal in treatment wetlands:

- The plant growth cycle seasonally stores and releases nitrogen, thus providing a "flywheel" effect for a nitrogen removal time series.
- The creation of new, stable residuals accrete in the wetland. These residuals contain nitrogen as part of their structure, and hence accretion represents a burial process for nitrogen.

On an instantaneous basis, plant uptake can be important for many wetland systems. A benchmark instantaneous growing season rate is suggested to be 120 gN/m²·yr (Kadlec, 2005d). The majority of the assimilated nitrogen is subsequently released during death and decay, but a small amount is permanently stored as new soil and sediment. The net removal of ammonia to accretion, via the vegetative cycle, is on the order of 10 gN/m²·yr. This amount is of great importance for very lightly loaded wetlands, but of no importance for heavily loaded systems.

The two forms of nitrogen generally used for assimilation are ammonia and nitrate nitrogen. Nitrate uptake by wetland plants is presumed to be less favored than ammonium uptake. But in nitrate rich waters, nitrate may become a more important source of nutrient nitrogen. Aquatic macrophytes utilize enzymes (nitrate reductase and nitrite reductase) to convert oxidized nitrogen to useable forms. The production of these enzymes decreases when ammonium nitrogen is present (Melzer and Exler, 1982). Plants such as cattails (*Typha latifolia*) are very able to utilize either nitrate or ammonia (Brix *et al*., 2002b), and so are algae (Naldi and Wheeler, 2002) and cultivated rice (Kronzucker *et al*., 2000). Dhondt *et al*. (2003) found that about half of the applied nitrate in a riparian wetland was utilized by plants, whereas half was denitrified.

In the Santee, California, study of a *Scirpus*/gravel HSSF wetland (Gersberg *et al*., 1984), the entire nitrate loss was ascribed to plant uptake in the absence of an exogenous carbon source and with essentially no ammonium in the nitrified influent. This process may also be important in other treatment wetlands. For instance, a short-term ^{15}N study of several SSF gravel wetland mesocosms (Zhu and Sikora, 1994) showed 70%–85% of the entire nitrate loss was plant uptake—in the absence of an exogenous carbon source and with essentially no ammonium in the nitrified influent. Different species responded differently: 70% of the nitrate was taken up by *Phragmites australis*, 75% by *Typha latifolia*, and 85% by *Scirpus atrovirens georgianus*. In the absence of definitive results on the proportions of nitrate versus ammonia uptake in treatment wetlands, some authors have opted to presume these are utilized in proportion to the quantities in the water (Martin and Reddy, 1997; Tanner *et al*., 2002a). However, process factors argue against this simple expectation. First, plants extract their nitrogen requirements via their root system, which is predominantly located in the wetland soil, with the possible exception of adventitious roots, which occur in the water column. Nutrients reach the subsurface root system via diffusion under appropriate circumstances, but more importantly via transpiration flux, the vertical water flow driven by the transpiration requirement of the plant (see Chapter 4). The upper soil horizon that contains the roots is typically anoxic and has a high carbon content, and therefore is capable of supporting denitrification (Crumpton *et al*., 1993). Nitrate that moves downward toward the root zone is therefore unlikely to survive in the same proportion as it exists in the water column above the soil.

The Effects of Vegetation Growth and Cycling

The removal of ammonia from water by wetland plants has been the subject of many studies (e.g., Reddy and DeBusk, 1985; Rogers *et al*., 1991; Busnardo *et al*., 1992; Tanner, 1996). Many such studies have been characterized by measurements of gross nitrogen uptake, with no deduction for subsequent losses due to plant death and decomposition, with the attendant leaching and resolubilization of nitrogen.

From the standpoint of nitrogen removal from wetland water, it is the net effect of the macroflora on water phase concentrations that is of interest. Here the terminology of Mueleman *et al*. (2002) will be used (see Figure 3.7):

- *Phytomass* refers to all vegetative material, living plus dead.
- *Biomass* refers to all living vegetative material.
- *Necromass* refers to all dead vegetative material.

The seasonal patterns of vegetation growth and nitrogen storage embody complex patterns of biomass allocation among plant parts, as well as the nitrogen content of those various portions of living and dead material. However, from the point of view of the annual ecosystem removal of nitrogen, uptake and return from the combination of biomass and necromass are the principal features of concern. On an annual average basis, the only concern is net removal to permanent storage. However, during the course of the year, uptake and return may occur at different times, thus influencing removals differently in different seasons. For these reasons, it is necessary to examine the transfers to and from the collective parts of the macrophytes, which is here defined as phytomass. During the course of the year, especially in temperate climates, phytomass increases during the growing season, and shrinks during the senescence season. The same pattern is followed by phytomass nitrogen.

A Mass Balance Framework

The purpose here is to make order-of-magnitude assessments of the role of vegetation in the overall set of ammonia nitrogen processes. This choice has the effect of establishing a "green and brown box," which interacts with the balance of the wetland ecosystem (see Figure 3.7). The nitrogen mass balance for that box is (instantaneously)

$$(J_u - J_r - J_b) = \frac{dN}{dt} \tag{9.34}$$

or for a fixed time period Δt:

$$(J_u - J_r - J_b)\Delta t = \Delta N \tag{9.35}$$

where

J_u = uptake of nitrogen by phytomass ($= U_e$), gN/m²·d
J_r = release of nitrogen from phytomass ($= L + D_a + D_b$), gN/m²·d
J_b = burial of nitrogen from phytomass ($= A_a + A_b$), gN/m²·d
dN/dt = storage change rate of nitrogen in phytomass, gN/m²·d
ΔN = increase in nitrogen storage in phytomass, gN/m²
Δt = time interval, d

The uptake of nitrogen is via the root system, which is usually belowground. Nitrogen must therefore be transported into the rhizosphere, by processes of diffusion (minor) and vertical movement driven by transpiration flux (major) (Reddy *et al*., 2005). Some of the new plant growth nutrient requirement is supplied by translocation from stores in the rhizomes, and some from uptake from pore water. It is possible that the presence of nitrogen-rich pore waters causes less withdrawal from rhizomes, and causes lesser storage in belowground tissues (Tanner, 2001a).

Nitrogen is returned to surface waters and pore waters by leaching and decomposition. It is likely that the majority of nitrogen in the necromass is returned, with lesser amounts transferred to permanent burial in the form of new soils and sediment. Over the course of a full calendar year, for a repetitively stable ecosystem, there is no change in the total phytomass, and $\Delta N = 0$. For that annual period, plant uptake is either returned (more) or buried (less). But, as can be seen from Figure 9.11, the total phytomass nitrogen grows in spring and early summer, and recedes in autumn. This annual cycle is more pronounced in cold climates, in response to the more pronounced seasonal conditions.

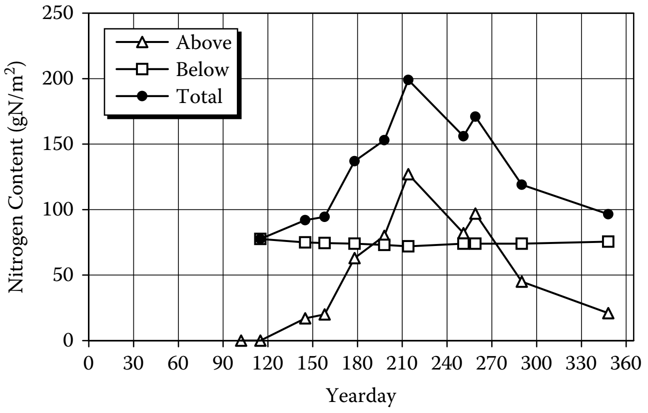


FIGURE 9.11 Seasonal patterns of nitrogen in *Phragmites australis* in the Netherlands for a fertilized stand. (Data from Mueleman *et al.* (2002) *Wetlands*, 22(4): 712–721.)

At this point in the development of knowledge about wetland plant nitrogen cycling, there is some good idea of the change in storage (ΔN) for a given time interval, but less about the three individual fluxes that lead to the storage (J_u, J_r, J_b).

A Speculative Numerical Assessment

The green and brown box, consisting of all phytomass nitrogen, expands during the growing season, and contracts during the balance of the year. The purpose here is to assess the approximate magnitude of these nitrogen withdrawals and returns upon the amount of ammonia nitrogen in the water column. Some useful insights may be gained by speculatively assigning uptake and burial (Kadlec, 2005b). These are:

1. A fixed proportion of the necromass nitrogen that returns to water.
2. A constant rate of burial (J_b) apportioned to the unfrozen season.
3. Nitrogen release driven by the amount of necromass during the unfrozen season.

As an order-of-magnitude illustration, an annual phytomass nitrogen cycle is presumed to follow a smoothed version of Figure 9.11. An annual accretion of 20 gN/m²·yr is proposed. This is apportioned over a growing season (unfrozen) of eight months, at a constant rate of 2.5 gN/m²·mo. Four times that amount, 80 gN/m²·yr, is presumed to be returned to water. Growth begins at the end of April, and ends in December, causing nitrogen uptake from April through August, totaling 156 gN/m². During September through December, 56 gN/m² is returned from senescing and decaying necromass from the current year. TN return is 80 + 56 = 136 gN/m² for the year, or 87% of the uptake. Only 13% of the nitrogen uptake finds its way into recalcitrant residual forms. However, during the spring growth period, the entire external nitrogen loading is consumed to create the standing crop. These seasonal effects are summarized in Figure 9.12. The loading to the wetland was 240–270 gN/m²·yr. Thus, it is seen that vegetative transfers make up major fractions of the external load.

Treatment wetland data show growing season vegetative uptakes of 20–100 gN/m², which occurs during a four- to six-month period in temperate climates. This results in growing season uptake rates of 40–200 g/m²·yr. A median benchmark uptake loading of 120 g/m²·yr has been selected here as a basis for evaluating external loadings. Examination of a large number of operational data sets for FWS wetlands leads to the conclusion that emergent and submergent plants are important contributors to the processing of ammonia in free water surface wetlands, for about half of the existing systems (Kadlec, 2005d). For instance, nitrogen storage in the roots and rhizomes in the inlet zone of a FWS *Phragmites/Typha* treatment wetland in Byron Bay, Australia, was 35 g/m²; in the leaves and stems it was 92 g/m² (Adcock *et al.*, 1995). Approximately 65% of the nitrogen added to this treatment wetland was found in the macrophyte biomass, due to low nitrogen loading (approximately 25–40 g/m²·yr).

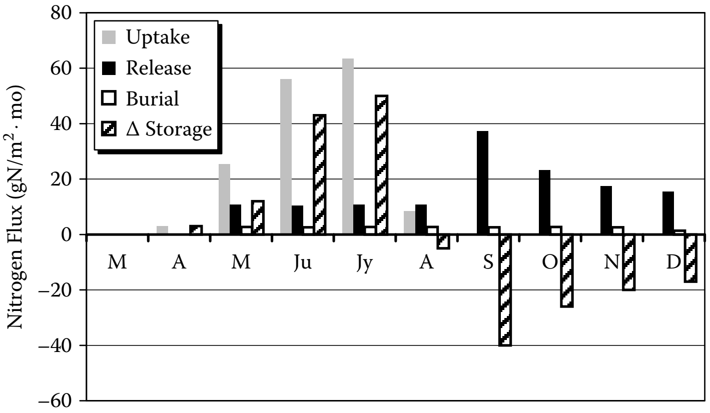


FIGURE 9.12 Hypothetical seasonal transfers of nitrogen corresponding to the measured growth pattern of Figure 9.11. The loading to the wetland was 240–270 gN/m²·yr. (Data from Mueleman *et al.* (2002) *Wetlands*, 22(4): 712–721.)

TABLE 9.6
Accretion Rates in FWS Wetlands

Location	Reference	Method	Water NH_4–N (Typical) (mg/L)	Accretion (cm/yr)	Nitrogen Burial (gN/m²·yr)
Louisiana	Rybczyk *et al.* (2002); 400–500 gC/gSoil; 2.0% N[a]	Feldspar	0.05	0.14	—
Michigan	Kadlec and Robbins (1984)	Lead 210	0.1	0.2	—
Everglades WCA2A	Reddy *et al.* (1991); 300–500 gC/gSoil; 3.0% N	Cesium 137	0.3	0.5	9
Everglades WCA2A	Craft and Richardson (1993a,b); 450 gC/gSoil; 3.2% N	Cesium 137	0.3	0.4	11.6
Everglades WCA3	Craft and Richardson (1993a,b); 450 gC/gSoil; 3.2% N	Cesium 137	0.1	0.3	10.7
Everglades, Florida	Robbins *et al.* (1999); 3.0% N[a]	Lead 210	0.3	0.5	11
Everglades, Florida	Chimney (2000), unpublished data; 500 gC/gSoil; 3.2% N	Feldspar	0.1	0.85	35
Iron Bridge, Florida	Miner *et al.* (2002)	Visual	0.1	1.17	—
Louisiana	Rybczyk *et al.* (2002); 400–500 gC/gSoil; 2.0% N[a]	Feldspar	15	1.14	23
Sacramento, California	Nolte and Associates (1998b); 4.3% N	Visual	16	1.5	44
Houghton Lake, Michigan	Kadlec (1997); 400–500 gC/gSoil; 3.2% N	Resurvey	10	1.8	56
Chiricahueto, Mexico	Soto-Jiménez *et al.* (2003); 10–40 gC/gSoil; 0.3% N	Lead 210	14	1.0	1.5

[a] Assumed value.

Accretion of Nitrogenous Residuals

The least studied aspect of nitrogen transfer in wetlands is in the creation of new soils and sediments, with their attendant nitrogen content. Not all of the dead plant material undergoes decomposition. Some small portions of both aboveground and belowground necromass resist decay, and form stable new accretions. Such new stores of nitrogen are presumed to be resistant to decomposition. The origins of new sediments may be from remnant macrophyte stem and leaf debris, remnants of dead roots and rhizomes, and from undecomposable fractions of dead microflora and microfauna (algae, fungi, invertebrates, bacteria).

The amount of such accretion has been quantified in only a few instances for free water surface wetlands (Reddy *et al.*, 1991; Craft and Richardson, 1993a,b; Rybczyk *et al.*, 2002), although anecdotal reports also exist (Kadlec, 1997a). Quantitative studies have relied upon either atmospheric deposition markers (radioactive cesium or radioactive lead), or introduced horizon markers, such as feldspar or plaster. Either technique requires several years of continued deposition for accuracy.

Reddy *et al.* (1991) used ^{137}Cs to estimate the rate of accretion in a mildly fertilized cattail wetland in Florida, which ranged from approximately 5 to 11 mm/yr of low bulk density material, less than 0.1 g/cm^3. The nitrogen content of these new accretions was measured to be approximately 3%, resulting in annual accretion rates of 11–24 gN/m^2·yr. Murkin *et al.* (2000) found 4.5–6.5 gN/m^2·yr annual accretion rates for low nutrient, mixed marshes in Manitoba. Soto-Jiménez (2003) reported net sedimentation of nitrogen of 11.3 gN/m^2·yr for a marsh receiving strong agricultural runoff. Hocking (1989b) estimated 8 gN/m^2·yr annual accretion rate for *Phragmites australis* in a nutrient-rich Australian setting. Klopatek (1978) estimated 5 gN/m^2·yr annual accretion rate for a *Schoenoplectus* (*Scirpus*) *fluviatilis* stand. Representative accretion rates are given in Table 9.6.

The manner of accretion has sometimes been presumed to be sequential vertical layering (Kadlec and Walker, 1999; Rybczyk *et al.*, 2002), but that view is likely to be overly simplified. At least two factors argue against simple layering: vertical mixing of the top soils and sediments (Robbins *et al.*, 1999), and the injection of accreted root and rhizome residuals at several vertical positions in the root zone. Nonetheless, new residuals are deposited on the wetland soil surface from various sources. The most easily visualized is the litterfall of macrophyte leaves, which results in top deposits of accreted material after decomposition. However, algal and bacterial processing which occurs on submersed leaves and stems results in litterfall and accretion of micro-detrital residuals.

Short-Term Anomalies

In addition to the considerations of long-term repetitive annual vegetation effects on wetland nitrogen processing, there are transient effects related to start-up of treatment wetlands. These transient events are different from the stable annual pattern of swelling and shrinking of the phytomass nitrogen storage. Results from transient studies must not be construed as being representative of long-term patterns. Some case study transient results are informative.

FWS Mesocosm Start-Up

Busnardo *et al.* (1992) operated FWS mesocosms vegetated with *Scheonoplectus* (*Scirpus*) *californicus*. The ammonia loading rates to the mesocosms were 330 and 670 gN/m^2·yr for two consecutive seven-month periods. Approximately 60% of the ammonia nitrogen removed was found in plant growth.

Although this experiment demonstrated that emergent macrophytes have the capacity to assimilate large quantities of ammonia, Busnardo *et al.* (1992) speculated that plants would have a lesser effect in mature wetlands.

SSF Mesocosm Start-Up

A number of studies in the literature focus upon newly planted mesocosms, which are monitored for performance during the subsequent period of plant development. For example, Rogers *et al.* (1991) reported on nitrogen processing in 25-L buckets filled with gravel and planted with *Schoenoplectus validus* rhizomes. Studies of ammonia removal commenced five weeks later, and continued for 35 weeks. Ammonia loading rates of 60–600 gN/m²·yr were applied over periods of 10–15 weeks. Removals ranged from 90–100%, of which about 90% was found in the vegetation. These rates of uptake are not counteracted by return fluxes, because no necromass was formed over the short duration of the tests. It was eventually found that the plants in the buckets remained in the colonizing mode for at least three years. (Rogers *et al.*, 1991).

Ammonia Loads to a New Wetland

Newly constructed wetlands are typically planted sparsely compared to the ultimate grow-out of vegetation. The development of the new vegetation creates a nitrogen demand that persists only during that grow-in period. For example, Sartoris *et al.* (2000) reported on the first two years of ammonia removal and plant coverage for a 9.9-ha FWS constructed wetland at Hemet, California. As the plant coverage went from near zero (planted clumps on 1.2-m spacing) to about 80% of *Schoenoplectus* spp., and the vegetation density increased by 67%, the ammonia load removed went from 98 down to 15 gN/m²·yr. Sartoris *et al.* (2000) concluded that plant uptake was most likely the primary sink for nitrogen during the two-year study. In this case of a FWS wetland, the increase in coverage by plants reduced the fraction of open water, and hence created a lesser potential for atmospheric reaeration to support nitrification.

Harvest to Remove Nitrogen

Nitrogen removal is theoretically possible via the harvest of plants and their associated nitrogen content. However, aboveground standing crops do not display a large potential for removal of nitrogen, even under the assumption that the entire crop could be recovered (Table 9.7). Based on the productivities given by DeBusk and Ryther (1987), potential nitrogen removal for floating large-leaved plants (*Eichhornia, Pistia, Hydrocotyle*) is in the range of 100–250 gN/m²·yr, and 50–150 gN/m²·yr for floating small-leaved plants (*Salvinia, Lemna, Spirodela, Azolla*).

Direct harvesting experience has shown that only a small fraction of the applied nitrogen can be recovered in harvested biomass (Table 9.7). Systems operating in tropical climates may be capable of greater sustained annual vegetative removals, which are enhanceable by harvest. Koottatep and Polprasert (1997) measured from 70 to 275 gN/m²·yr, depending upon harvesting frequencies ranging from no harvest to every eight weeks, respectively.

Harvest may involve complete removal in the case of floating plants (*Lemna minor*, *Eichhornia crassipes*), or cutting of aboveground parts of rooted plants such as *Typha*, *Schoenoplectus*, and *Phragmites*. Harvesting typically requires expensive mechanical equipment, and is labor-intensive for large systems. For instance, a one-time harvest of floating mats of *Typha* in a Florida treatment wetland cost approximately $16 per cubic meter of wet material, or about $8 per kilogram of nitrogen removed. However, in the small SSF systems, such as those commonly found in

TABLE 9.7
Amount of Nitrogen in the Standing Aboveground Stock Compared to Nitrogen Loadings

Location	Reference	Type	Nitrogen Stock (gN/m²)	Applied Nitrogen (gN/m²·yr)	Percent Removable
Ondrejov, Czech Republic	Vymazal *et al.* (1999)	SSF *Phragmites*	51	1,183	4.3
Kolodeje, Czech Republic	Vymazal *et al.* (1999)	SSF *Phragmites*	20	493	4.1
Chmelna, Czech Republic	Vymazal *et al.* (1999)	SSF *Phalaris*	26.5	1,397	1.9
Zasada, Czech Republic	Vymazal *et al.* (1999)	SSF *Phalaris*	—	—	0.8
Hamilton, New Zealand	Tanner (2001a)	SSF *Schoenoplectus*	23	431	5.3
Hamilton, New Zealand	Tanner (2001a)	SSF *Schoenoplectus*	40	1,256	3.2
Sacramento, California	Nolte and Associates (1998b)	FWS *Typha* + *Scirpus*	60	360	16.5
ENR, Florida	Everglades ENR Cell 1, unpublished data	FWS *Typha*	4.7	8	60
Byron Bay, Australia	Adcock *et al.* (1995)	FWS *Leersia* + *Urochloa*	44	203	21
Houghton Lake, Michigan	Houghton Lake, Michigan–based 50 ha, unpublished data	FWS *Typha*	25	10	250
Malham, United Kingdom	Hurry and Bellinger (1990)	FWS *Phalaris*	49	469	11
Duplin County, North Carolina	Hunt *et al.* (2002)	FWS *Typha*	32	392	8
Duplin County, North Carolina	Hunt *et al.* (2002)	FWS *Scirpus*	35	420	8
Greensboro, North Carolina	Hunt *et al.* (2002)	FWS *Typha*	20	971	2

Europe, harvesting is easy and forms a negligible amount within the annual O&M costs.

The problem of biomass disposal is often not easily resolved. Harvested biomass may either be composted, or digested to form a biogas product. Composting requires transportation costs, and dedicated land area. Biogas generation from water hyacinths has been shown to be feasible (Biljetina *et al*., 1987; Joglekar and Sonar, 1987); however, sludge disposal remains a problem. The capital cost of harvesting and gas generation about is about the same as for the rest of the wastewater treatment plant, and is thus prohibitively expensive (Chynoweth, 1987). As a consequence of these difficulties, plant harvesting is not favored for nitrogen removal (Crites and Tchobanoglous, 1998), and has seldom been used except for floating plants.

Soil and Sediment Effects on Nitrogen Processing

Apart from accretion, wetland solids form a large pool of nitrogen, some of which is available for exchange with surface waters and pore waters. As noted above, sorption and cation exchange are active processes in the wetland environment. These nitrogen solid storages will stabilize under continuous operation of a treatment wetland, but are nonetheless active, and exchange compounds with their surroundings. Thus the image of nitrogen compounds traveling with the flowing water is incorrect; nitrogen follows a "park and go" trajectory through the wetland.

Kadlec *et al*. (2005) reported these exchanges for SSF treatment wetlands. Four mesocosm trains and one field-scale wetland contained well-established bulrushes (*Schoenoplectus tabernaemontani*), and another field-scale wetland remained unvegetated. The systems were operated at steady inflows, with a nominal detention times of four to five days. The incoming ammonium nitrogen ranged from 18.5–177 g/m^3, and removals ranged from 15% to 90% for the various feed waters. Each system was dosed with a single pulse of ^{15}N ammonium mixed into the feed wastewater, and the fate and transport of the isotopic nitrogen were determined. The ^{15}N pulses took 120 days to clear the heavily loaded field-scale wetlands. During this period small reductions in ^{15}N were attributable to nitrification/denitrification, and a larger reduction due to plant uptake. Mesocosm tests ran for 24 days, during which only 1–16% of the tracer exited with water, increasing with nitrogen loading. Very little tracer gas emission was found, about 1%. The majority of the tracer was found in plants (6–48%) and sediments (28–37%). These results indicated a rapid absorption of ammonium into a large sediment storage pool, of which only a small proportion was denitrified during the period of the experiment. Plant uptake claimed a fraction of the ammonium, determined mainly by the plants requirement for growth rather than the magnitude of the nitrogen supply. A rapid return of ammonium to the water was also found, so that movement of ^{15}N through the wetland mesocosms comprised a "spiral" of uptake and release along the flow path.

9.5 NITROGEN MASS BALANCES

The individual process considerations discussed above may be combined to form the integrated concept of nitrogen fluxes in treatment wetlands. This interpretive step is very important, because it

1. Identifies the true rates of ammonification, ammonia oxidation, and denitrification.
2. Places the role of the vegetative nitrogen cycle in the context of the microbial processes.
3. Allocates the fate of added nitrogen to storage, leakage, and gasification.

The use of the percent removal measure may be very misleading for separate nitrogen species. For example, U.S. EPA (1993f) found that approximately half of the SSF wetlands inventoried had negative percent removals for ammonia. In the absence of speciated nitrogen mass balances, that technology assessment ascribed the good performance to lack of algae, oxygen availability and long detention, and poor performance to short rooting depth and oxygen deficiency. However, in the absence of adequate data on ammonification, U.S. EPA (1993f) dismissed that process as not being a contributing factor. Much more information is now available, and it is possible to examine the nitrogen interconversions in more detail.

Mass Balance Case Studies

Only a few wetland studies have reported mass balances for the interrelated species of nitrogen (Tanner and Kadlec, 2002; Senzia *et al*., 2002b; Bishay and Kadlec, 2005; Kadlec *et al*., 2005). In all cases, the involvement of vegetation in the nitrogen cycle is somewhat speculative, because it depends upon estimates of biomass and tissue nitrogen content. Nonetheless, much is known about standing stocks and turnover rates, as well as the (narrow) bounds on nitrogen percentages in that biomass. Here three examples of FWS wetland nitrogen mass balances will be explored: (1) a lightly loaded polishing wetland, (2) a leaky wetland treating contaminated river water, and (3) a seasonal wetland treating nitrogenous mine wastewaters. In each case, long-term performance is examined, and consequently seasonal effects are not elucidated. One example of mass balance for an HSSF wetland is presented as well.

Orlando Easterly, Florida, FWS Wetland

This treatment wetland has been in operation since 1987, and is described in general terms in U.S. EPA (1993a). It is a 494-ha constructed free water surface wetland with 17 compartments in a series and parallel arrangement, which receives about 60,000 m^3/d of highly treated municipal effluent. The cells were vegetated with soft-tissue emergent plants, and the vegetative communities evolved over time to a mixed marsh condition. In addition to annual and specialty project reports,

TABLE 9.8A
Average Inlet and Outlet Concentrations for the Orlando Easterly, Florida, FWS Wetland for 1993–2002

Parameter	Inlet (mg/L)	Outlet (mg/L)	Mean (mg/L)	Fraction	Assumed Rain (mg/L)
HLR, cm/d	1.17	1.15	—	—	0.41
Organic N	1.67	0.98	1.32	—	1.0
Ammonia N	0.33	0.14	0.23	0.545	0.5
Nitrite N	0.60	0.04	0.32	—	—
Nitrate N	0.26	0.13	0.19	0.455	0.5
TSS	1.2	4.0	—	—	—
$CBOD_5$	2.0	2.6	—	—	—
DO	6.1	8.9	—	—	—
Alkalinity	94	92	—	—	—

Source: City of Orlando operating data.

there have been several published papers (Jackson, 1989; Jackson and Sees, 2001; Martinez and Wise, 2003a,b; Wang *et al*., 2006a,b). Data used here are from the ten-year period 1993–2002.

Nitrogen totals less than 3 mg/L entering the system, and less than 1.4 mg/L in the effluent from the wetland. Atmospheric contributions are not negligible under these circumstances, and are estimated at 2.0 mg/L based upon other Florida data. The inlet hydraulic loading was 1.2 cm/d, and rainfall averaged about 0.4 cm/d (Table 9.8A). Particulate nitrogen is not a factor, because the TSS content of the incoming water is very low (1.2 mg/L). The data combine to produce a TN inlet loading of 11.3 gN/m^2·yr, apportioned across the species as indicated in Figure 9.13. This is much less than the required nitrogen for even modest plant growth, indicating that the vegetative cycle must draw upon internal sources of nitrogen. There was net removal of all forms of nitrogen, summing to a 70% reduction in the load of TN. The inlet–outlet concentration reduction was less, 55%, because it does not include the contribution of rainfall nitrogen.

Since measurements were not made of vegetative nitrogen processes, assumptions must be made. The wetland was moderately well vegetated, with some open water, leading to the assumption of an annual productivity of 1,000 g dw/m^2·yr with an assumed nitrogen content of 2%. Of this, 10% was assumed to be buried as new sediments (Table 9.8B). Both nitrate and ammonia were presumed to be used to support growth, in proportion to their availability in the water. Average concentrations were used to determine the uptake ratio, although selective spatial utilization may have occurred.

This information is adequate to calculate all the average annual transfers within the wetland via mass balances. The pattern of nitrogen transfers is dominated by the vegetative cycle (Figure 9.13). Production of ammonia from decomposition of biomass is eight times higher (20.65 gN/m^2·yr) than the reduction in ammonia in the water from inlet to outlet (2.64 gN/m^2·yr). Nitritation is seven times higher than the reduction in the flowing ammonia load (11.32 versus 1.57 gN/m^2·yr), and that high internal load of nitrite is subsequently nitrified to nitrate. Some nitrate is lost through denitrification, but more is used to support

TABLE 9.8B
Assumptions for the Orlando Easterly Wetland Carbon and Oxygen Supplies

Assumption			Notes
Biomass produced =	1,000	g dm/m^2·yr	—
Carbon content =	0.5	—	50%
Useable carbon fraction =	0.3	—	30%
Carbon available =	150	g/m^2·yr	—
Denitrification carbon requirement =	140	g/m^2·yr	1.07 × N
Biomass N uptake =	20	g/m^2·yr	2% N
Biomass buried =	100	g dw/m^2·yr	10%
Nitrogen buried =	2.0	g/m^2·yr	2% N
Oxygen needed =	36	g/m^2·yr	3.43 × nitritation + 1.14 × nitrification, plus DO increase
Daily oxygen needed =	0.16	g/m^2·d	—

Note: Biomass is the assumed source of carbon, and oxygen requirements are determined from Figure 9.14 fluxes.

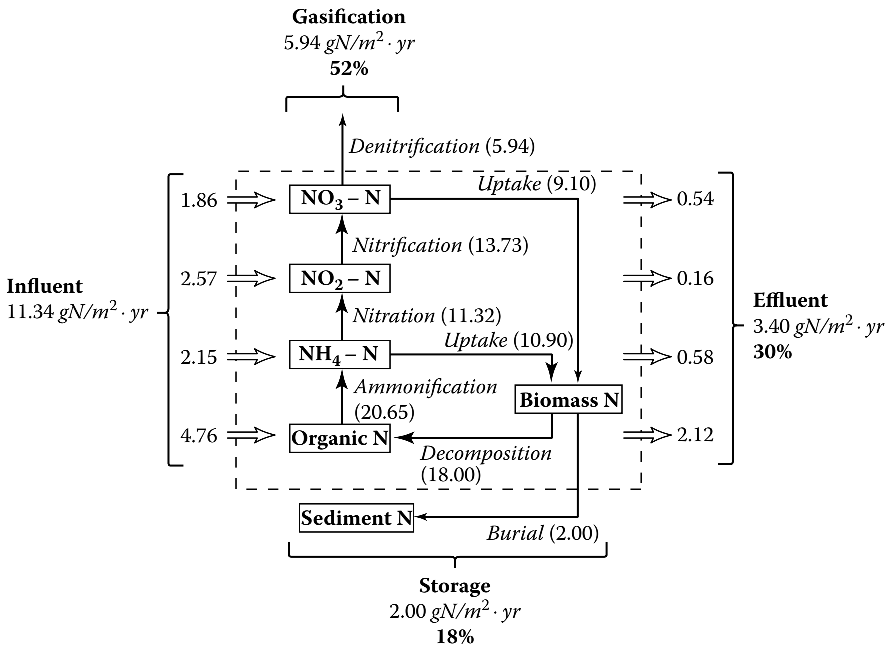


FIGURE 9.13 Estimated annual nitrogen fluxes in the Orlando Easterly treatment wetland ($gN/m^2 \cdot yr$). The vegetation cycle dominates this lightly loaded system.

plant growth. However, denitrification amounts to 52% of the net nitrogen input, whereas accretion of new sediments represents only 18%.

The required supplies of ancillary chemicals were present in the wetland (Table 9.8 A, B). Dissolved oxygen is present to support ammonia oxidation and the observed reaeration, which is calculated to need 0.16 $gO/m^2 \cdot d$, well within the range of expected atmospheric reaeration (see Chapter 5). The required alkalinity is also available to support ammonia oxidation. There is no carbon in the inlet water

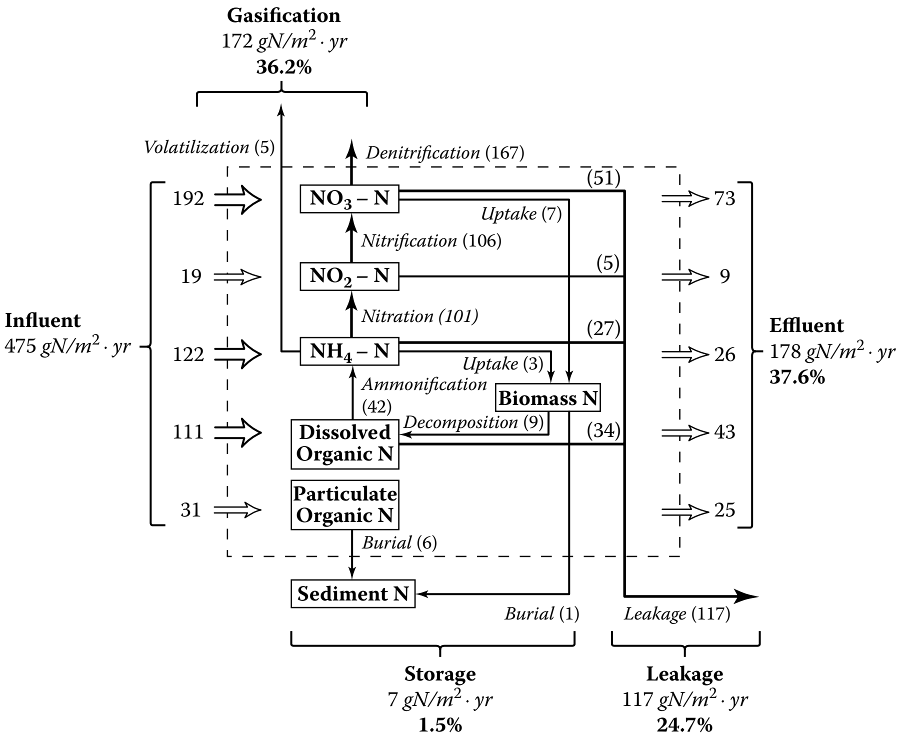


FIGURE 9.14 Estimated annual nitrogen fluxes in the Imperial, California, demonstration wetland ($gN/m^2 \cdot yr$). The vegetation cycle is of little importance in this system.

TABLE 9.9A
Average Inlet and Outlet Concentrations for the Imperial, California, FWS Wetland for 2001–2004

Parameter	Inlet (mg/L)	Outlet (mg/L)	Leakage (mg/L)	Fraction
HLR, cm/d	19.3	12.5	6.8	—
Dissolved organic N	1.77	1.01	1.39	—
Particulate organic N	0.46	0.60	0.53	—
Ammonia N	1.64	0.53	1.09	0.344
Nitrite N	0.25	0.18	0.22	—
Nitrate N	2.65	1.49	2.07	0.656
TSS	179	11	—	—
$CBOD_5$	7.2	5.2	—	—
DO	8.2	7.2	—	—
Sulfate S	661	618	640	—
Alkalinity	239	203	—	—

Source: Imperial Irrigation District operating data.

to support denitrification ($CBOD_5$ = 2 mg/L), but the biomass cycle produces enough available carbon to fuel heterotrophic denitrifiers.

Imperial, California, FWS Wetland

This FWS treatment wetland system has been in operation since 2000, and the data used here are from the four-year period 2001–2004. It consists of a 3.88-ha sedimentation basin, followed by 4.72 ha in four wetland cells in series. The system received 16,600 m³/d of agricultural runoff. The cells are about 75% open water and 25% vegetated with bulrushes. Data were summarized in Tetra Tech, Inc. (TTI) (2006). The TN areal loading was over 40 times that at the Orlando Easterly Wetland.

The hydraulic loading to the system is high (19.3 cm/d), and 35% infiltrates. The incoming water has high TSS (179 mg/L, Table 9.9A), which is effectively removed in the sedimentation basin and wetland cells. However, particulate nitrogen is low, and is not reduced in the system. Oxidized and dissolved organic nitrogen dominate the inflow, which has a TN of 6.8 mg/L; the outflow has 3.8 mg/L TN (44% concentration reduction) (Table 9.9A). About 25% of the nitrogen load is infiltrated (Figure 9.14). In contrast to the Orlando system, the vegetative cycle at Imperial has almost no effect on the nitrogen budget. Vegetation was sparse, and gross uptake was estimated to be only 2% of the incoming nitrogen load.

Ammonification primarily reduces the load of dissolved organic nitrogen. Nitrification and denitrification dominate the processing matrix (Figure 9.14). The required supply of oxygen, in excess of the observed depletion of the water column DO, was 1.45 gO/m²·d, which is reasonably within the range of expected atmospheric reaeration (see Chapter 5) (Table 9.9B). Sufficient alkalinity was present to support nitrification. However, there was estimated to be not enough carbon available from the decomposition of the sparse vegetation, or incoming $CBOD_5$, to support denitrification. A possible candidate mechanism was sulfur-driven autotrophic denitrification. The incoming water contained over 600 mg/L of sulfate. If only a small fraction, less than 1%, of this were reduced to sulfide in the wetland sediments, then that sulfide could have supported the balance of the observed denitrification over carbon-driven, heterotrophic denitrification.

Musselwhite, Ontario, FWS Wetland

The Musselwhite gold mine uses FWS wetland treatment to deal with the ammonia that is produced in the gold extraction and cleanup processes. This 2.5-ha constructed wetland was operated in the unfrozen seasons, at a depth of about 30 cm and a hydraulic loading rate of 50 cm/d (Bishay and Kadlec, 2005). The site was a former forested peatland, with the trees cut down, and logs and brush left in the wetland. Marsh vegetation consisted of *Equisetum* spp., *Typha* spp., and *Carex* spp.

TABLE 9.9B
Assumptions for the Imperial, California, FWS Wetland Carbon and Oxygen Supplies

Assumption			Notes
Biomass produced =	500	g dw/m²·yr	—
Carbon content =	0.5	—	50%
Useable carbon fraction =	0.3	—	30%
Carbon available =	75	g/m²·yr	—
Denitrification carbon requirement =	179	g/m²·yr	1.07 × N
Denitrification sulfide requirement =	164	g/m²·yr	1.69 × N excess
Sulfate incoming =	47,000	g/m²·yr	—
Biomass N uptake =	10	g/m²·yr	2% N
Biomass buried =	0.1	g dw/m²·yr	10%
Nitrogen buried =	1.0	g/m²·yr	2% N
Oxygen needed =	529	g/m²·yr	3.43 × nitritation + 1.14 × nitrification, less DO reduction
Daily oxygen needed =	1.45	g/m²·d	—

Note: Biomass is the assumed source of carbon, and oxygen requirements are determined from Figure 9.15 fluxes.

TABLE 9.10A
Average Inlet and Outlet Concentrations for the Musselwhite, Ontario, FWS Wetland for 1997–2002

Parameter	Inlet (mg/L)	Outlet (mg/L)	Mean (mg/L)	Fraction
HLR, cm/d	—	52	—	—
Dissolved organic N	1.18	0.75	0.97	—
Ammonia N	11.61	3.18	7.40	0.344
Nitrite N	0.85	0.19	0.52	—
Nitrate N	5.79	10.10	7.95	0.656
TSS	5.0	5.0	—	—
DO	10.7	2.8	—	—
Alkalinity	124	86	—	—

Source: Data from Bishay and Kadlec (2005) In *Natural and Constructed Wetlands: Nutrients, Metals, and Management*. Vymazal (Ed.), Backhuys Publishers, Leiden, The Netherlands, 176–198.

Water is stored over winter in a pond, and is essentially devoid of TSS and BOD. However, partial nitritation and nitrification take place in the storage pond, leading to a mix of the nitrogen species entering the wetland (Table 9.10A, B).

The TN areal loading was over 300 times that at the Orlando Easterly wetland. Therefore, the vegetation utilization of nitrogen is of negligible consequence (Figure 9.15). There was also little organic nitrogen entering the wetland, and as a result dissolved inorganic nitrogen dominates the set of transfer processes. There was 75% reduction in the ammonia concentration, which is the regulatory parameter of interest. Because of nitrification, there was an increase in the nitrate concentration though the wetland of 80%, and these two effects partially counteract in TN reduction (25%).

Two anomalies were present concerning the supplies of ancillary chemicals. First, if nitritation and nitrification were purely heterotrophic, the conventional chemistry indicates a need for 20.2 gO/m^2·d, of which 4.1 was supplied by a depletion of incoming DO (Bishay and Kadlec, 2005). The net requirement of 16.1 gO/m^2·d is well outside the range of expectations for reaeration. Second, the carbon supply for purely heterotrophic "conventional" denitrification would be ten times higher than that estimated to be available from biomass decomposition.

An alternative possibility is that autotrophic nitrification/denitrification could have occurred. Van Loosdrecht and Jetten (1998) note that "autotrophic nitrifiers might be responsible for a range of 'strange' nitrogen conversions in wastewater treatment processes." The presence of considerable nitrite in the inlet water (13% of oxidized nitrogen), as well as ammonia, created conditions conducive for Equation 9.31. This relieves both the oxygen and carbon requirements, by about half (Bishay and Kadlec, 2005). The transfers in Figure 9.15 reflect this assumption.

Dar es Salaam, Tanzania, HSSF Wetland

This HSSF wetland system is used to provide secondary treatment of effluent from a primary facultative pond at the University of Dar es Salaam, Tanzania (Senzia *et al.*, 2002b). The system consists of four HSSF wetland beds in parallel; each bed is 40.7 m^2, and the hydraulic loading was approximately 5 cm/d. Nitrogen in the pond effluent is dominated by ammonia, and by organic nitrogen (Figure 9.16). The influence of plant biomass cycling is apparent; a large fraction of the influent ammonia (32%) is uptaken by the plants; the majority of this is returned back to the system as organic nitrogen (plant biomass increases the influent organic-nitrogen loading by 46%). However, organic nitrogen undergoes ammonification and this nitrogen is returned to the ammonia pool. Nitrification and denitrification are significant, exporting 48.8% of the applied nitrogen load; however, the majority of the nitrogen present in the effluent is in the form of ammonia (88% of the effluent nitrogen), and the export of effluent nitrogen accounts for 46.4% of the influent load. Only 4.8% of the nitrogen is stored in sediments and plant detritus.

TABLE 9.10B
Assumptions for the Musselwhite, Ontario, FWS Wetland Carbon and Oxygen Supplies

Assumption			Notes
Biomass produced =	600	g dw/m^2·yr	—
Carbon content =	0.5	—	50%
Useable carbon fraction =	0.3	—	30%
Carbon available =	90	g/m^2·yr	—
Heterotrophic denitrification supported =	84	g/m^2·yr	1C/1.07
Autotrophic denitrification =	864	g/m^2·yr	Difference
Biomass N uptake =	12	g/m^2·yr	2% N
Biomass buried =	60	g dw/m^2·yr	10%
Nitrogen buried =	1.2	g/m^2·yr	2% N
Oxygen needed =	2,167	g/m^2·yr	1.6 × nitritation + 3.0 × nitrification
Daily oxygen needed =	5.9	g/m^2·d	—
Biomass produced =	600	g dw/m^2·yr	—

Note: Biomass is the assumed source of carbon, and oxygen requirements are determined from Figure 9.16 fluxes.

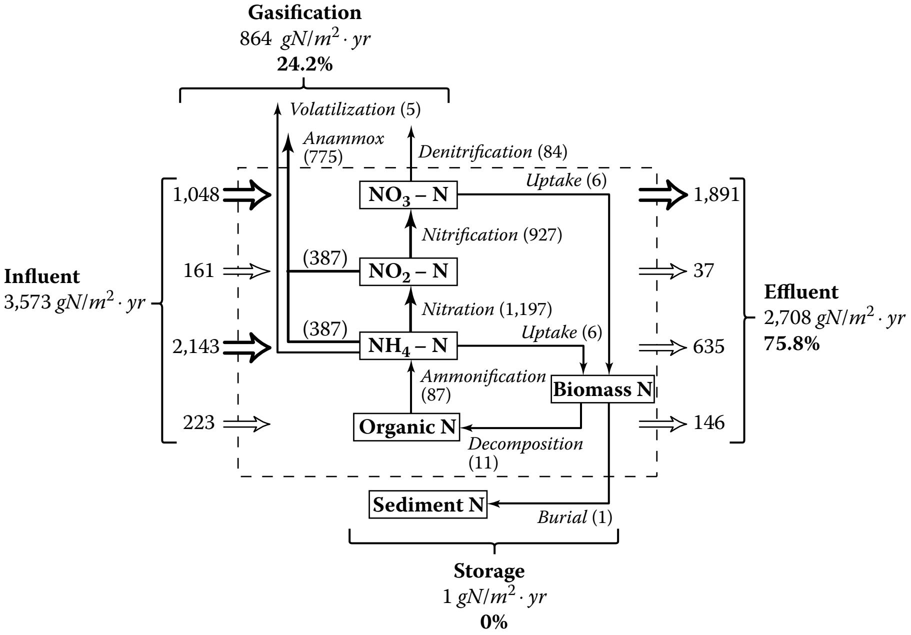


FIGURE 9.15 Mass balance for nitrogen flows in the Musselwhite, Ontario, FWS wetland ($gN/m^2·yr$), for an autotrophic nitrification/denitrification assumption. Base data were means of six years' measurements. The rate of denitrification, 84 $gN/m^2·yr$, was estimated based upon carbon availability. (Adapted from Bishay and Kadlec (2005) In *Natural and Constructed Wetlands: Nutrients, Metals, and Management.* Vymazal (Ed.), Backhuys Publishers, Leiden, The Netherlands, pp. 176–198.)

Figure 9.16 is an excellent illustration of the pitfalls of using input–output analysis for specific nitrogen species. If ammonia is considered to the exclusion of other nitrogen species, one could conclude that the system is not particularly effective in ammonia-nitrogen removal (influent of 326 $gN/m^2·d$; effluent of 217 $gN/m^2·d$). This of course ignores the impacts of the organic nitrogen fraction and the importance of plant biomass cycling in this system. Only when all of the nitrogen species are considered in concert can an overall understanding of nitrogen removal be developed.

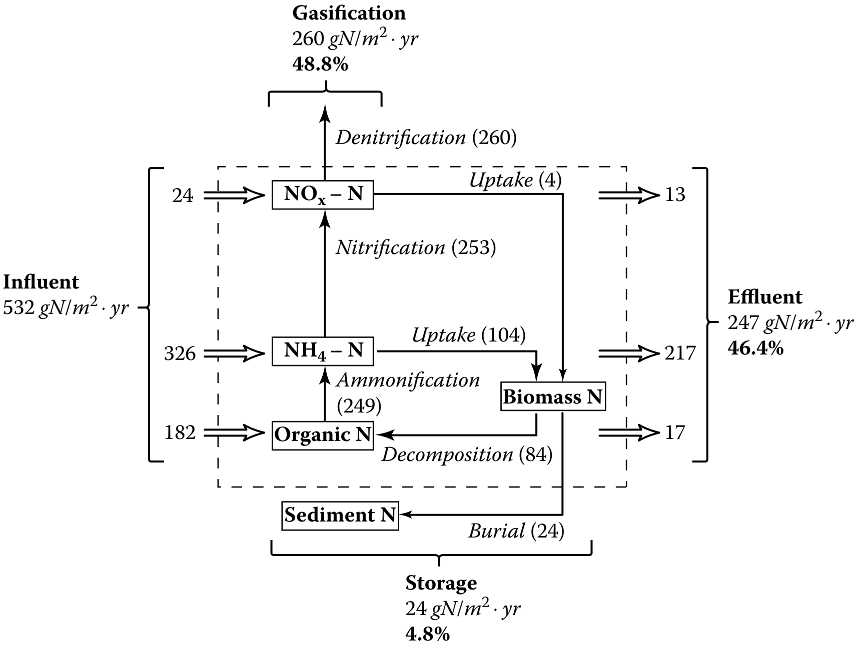


FIGURE 9.16 Nitrogen species mass balances for a *Phragmites mauritius* HSSF wetland. (Adapted from Senzia *et al.* (2002b). *Modeling nitrogen transformation in horizontal subsurface flow constructed wetlands planted with* Phragmites mauritius. Mbwette (Ed.). Proceedings of the 8th International Conference on Wetlands Systems for Water Pollution Control, 16–19 September 2002; Comprint International Limited: University of Dar es Salaam, Tanzania, pp. 813–827.)

Implications of the Nitrogen Mass Balance Network

A few important points emerge from this integrated view of nitrogen processing. First, the magnitude of the vegetative nitrogen cycle is by no means always trivial, because uptake can represent a good portion of the net removal for lightly loaded systems. However, net burial is only a fraction of plant uptake. Second, the influence of the biomass decay causes the true amount of ammonification to exceed the apparent rate based only on water analyses. Third, the true amount of nitrification greatly exceeds the amount based only on ammonia input–output water analyses. A sequential nitrogen kinetic model corrects for the production of ammonium from organic nitrogen, and calibrates to have higher rate constants accordingly. Finally, the rate of denitrification far exceeds the rate based only on nitrate input–output water analyses. The contribution of nitrification means that apparent denitrification is much smaller than the true value.

When microbial processes dominate, and the effects of the vegetative cycle are negligible, there are three independent mass balances that may be contrived without influences from other nitrogen species: (1) organic nitrogen, (2) TKN, and (3) TN. These are all groups of compounds, not single chemical entities. The overall reactions are:

$$\text{Organic N} \rightarrow NH_3 \tag{9.36}$$

$$\text{TKN} \rightarrow products_1 \tag{9.37}$$

$$\text{TN} \rightarrow products_2 \tag{9.38}$$

where

$products_1$ = oxidized N plus gases (NH_3, N_2O, N_2)
$products_2$ = gases (NH_3, N_2O, N_2)

Accordingly, it is reasonable to write disappearance models for these three, without including any production terms. There is, however, a background concentration of organic nitrogen (C^*), which influences all three rates. Nitrate, nitrite, and ammonia are all produced as well as consumed in the conversion web, and therefore reaction kinetics for these are of necessity more complex.

9.6 PERFORMANCE FOR ORGANIC NITROGEN

Organic nitrogen is present in domestic and municipal effluents. Wetlands typically receive these wastewaters after partial treatment, and the wetland influent then contains varying amounts of the original organic nitrogen, depending upon the type of pretreatment. Wetlands are themselves organic-rich sites, with considerable internal production of nitrogenous compounds. Incoming organic nitrogen is reduced, but not below the background concentration created by residuals and wetland return fluxes. Organic nitrogen is rarely, if ever, a regulated water quality parameter.

Loading Considerations

Measurements of ammonification rates in natural wetlands ranged from 1 to 15 g/m^2·yr (annual average 1.5) in a swamp forest in central Minnesota (Zak and Grigal, 1991) and from 4.3 to 5.9 g/m^2·yr in a Minnesota bog (Urban and Eisenreich, 1988). Treatment wetlands are typically nutrient-enriched environments, and process more organic nitrogen than natural systems.

Reduction of Organic Nitrogen in FWS Wetlands

The median net period-of-record removal rate for 60 FWS systems receiving more than 5 mg/L of organic nitrogen is 90 g/m^2·yr (Table 9.11). There is, however, wide variability among systems.

As detailed in Chapter 6, it is possible to represent annual wetland performance as the effluent concentration produced (C_o) by a given loading rate in (LRI = HLR × C_i) and concentration (C_i). In the broad context, multiple data sets are represented by a trend that shows increasing C_o with increasing LRI, with different groupings associated with each inlet

TABLE 9.11
Annual Reduction of Organic Nitrogen in FWS Wetlands

Stipulations

1. Data restricted to wetlands receiving inlet $C > 5$ mg/L organic nitrogen.
2. Period of record averages are used in calculations.
3. For k-value calculations, the following P-k-C^* parameters are selected:
   a. $C^* = 1.5$ mg/L
   b. $P = 3$ TIS
4. Ranges of variables:

	HLR (cm/d)	OGN In (mg/L)	OGN Out (mg/L)
Mean	6.3	18.2	8.1
Median	3.9	10.8	5.7
Max	49.9	69.5	29.6
Min	1.0	5.7	1.6

Results (N = 60 wetlands)

Percentile	Load Removed (g/m^2·yr)	Rate Coefficient (m/yr)
0.0	3	0.6
0.1	28	5.0
0.2	46	7.9
0.3	55	10.7
0.4	78	14.6
0.5	90	17.3
0.6	131	19.7
0.7	180	27.4
0.8	264	36.2
0.9	395	61.9
1.0	3,461	262.4

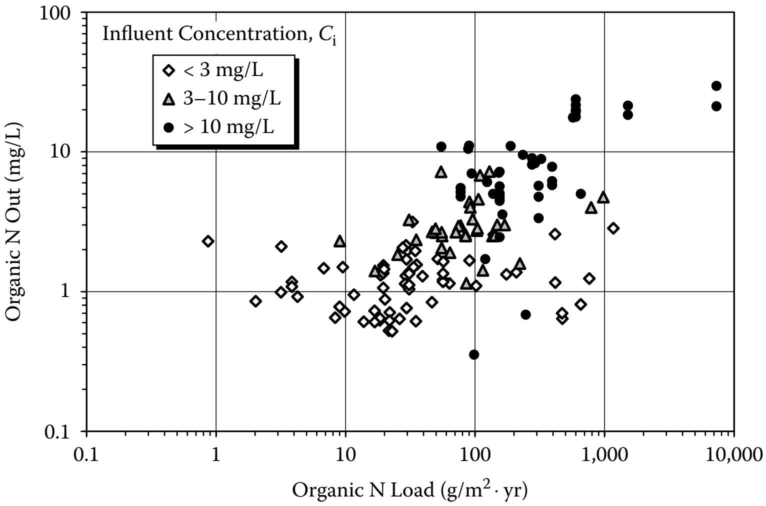


FIGURE 9.17 Load–concentration plot for organic nitrogen in FWS wetlands. Points are separated according to the inlet concentration range. Each point represents the entire period of record (POR) for one of 147 wetlands.

concentration (Figure 9.17). The overall slope of the intersystem data is approximately 0.5 on the log–log coordinates but is close to 1.0 in the central loading region. However, if the data are sorted into different inlet concentration ranges, a different picture emerges. For inlet concentrations in the range of 0.5–2.5 mg/L, there is little change in the outlet concentrations as the organic nitrogen loading is varied. Importantly, if hydraulic loading is reduced at constant inlet concentration, there is far less effect than indicated by the 0.5 slope of the overall data trend. Loading is an insufficient design specification because hydraulic load and inlet concentration are not interchangeable factors in the load representation.

Reduction of Organic Nitrogen in HSSF Wetlands

Many studies of HSSF wetlands have ignored the impact of organic nitrogen, even though ammonification of organic nitrogen represents a potential route of ammonia production within HSSF wetlands beds (Wallace and Knight, 2006; WERF database, 2006). Annual average effluent concentrations as a function of influent organic nitrogen loading for 123 HSSF wetlands (198 system-years of data) are summarized in Figure 9.18.

As seen in Figure 9.18, it is seen that there is a trend towards increasing effluent concentrations with increasing influent loadings of organic nitrogen, with an overall slope

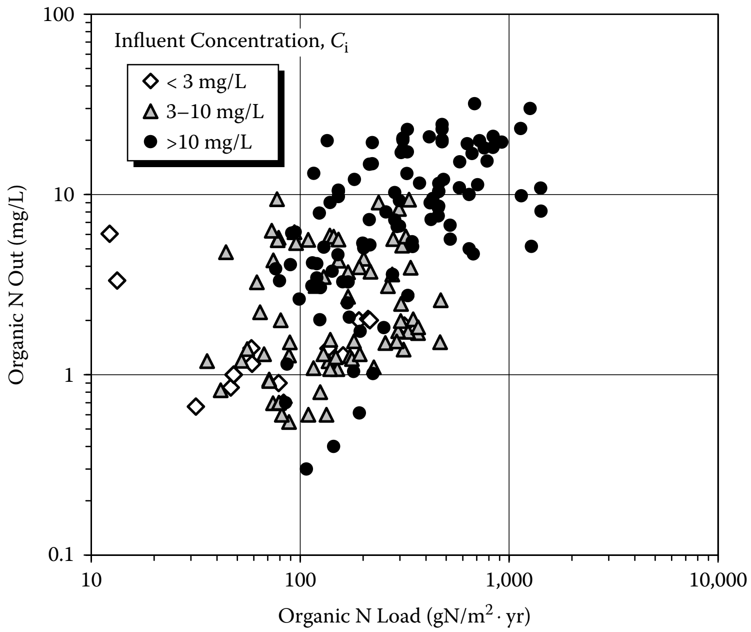


FIGURE 9.18 Outlet organic nitrogen as a function of inlet organic nitrogen loading for HSSF wetlands. Data are annual averages for 198 wetland-years from 123 wetland cells.

TABLE 9.12
Annual Reduction of Organic Nitrogen in HSSF Wetlands

Stipulations

1. The decomposition of 2000 g/m²·yr of biomass causes production of 36 gN/m²·yr of organic nitrogen.
2. Annual averages are used in calculations.
3. For *k*-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 1.0$ mg/L
   b. $P = 6$ TIS
4. Ranges of variables:

	HLR (cm/d)	OGN In (mg/L)	OGN Out (mg/L)
Mean	7.0	13.0	6.8
Median	4.7	11.6	4.3
Max	41.2	107.3	73.2
Min	0.1	0.0	0.0

Results ($N \cdot t = 190$ wetland-years)

Percentile	Load Removed (g/m²·yr)	Rate Coefficient (m/yr)
0.05	27	1.9
0.10	37	3.8
0.20	57	6.5
0.30	79	8.8
0.40	95	12.4
0.50	112	19.6
0.60	137	25.6
0.70	158	38.0
0.80	194	72.4
0.90	249	124.2
0.95	305	168.3

of the intersystem data set of approximately 1.0 on log–log coordinates. However, when the influent loadings are broken down by concentration ranges, it is apparent that this relationship does not hold for systems with $C_i < 3$ mg/L, presumably because these systems are operating at an influent concentration close to the background concentration (C^*). Furthermore, there is considerable variability among systems. The median annual average removal of organic nitrogen is 112 g/m²·yr, as summarized in Table 9.12.

Background Concentrations of Organic Nitrogen

Treatment wetlands data display decreases in organic nitrogen with contact time, which are consistent with first-order reduction kinetics, but show a nonzero background concentration. For long detention times, corresponding to large distances from the inlet, small concentrations of organic N persist. Those background concentrations typically are in the range of 0.5–2.0 mg/L, and are therefore nontrivial with respect to some regulatory requirements for TN.

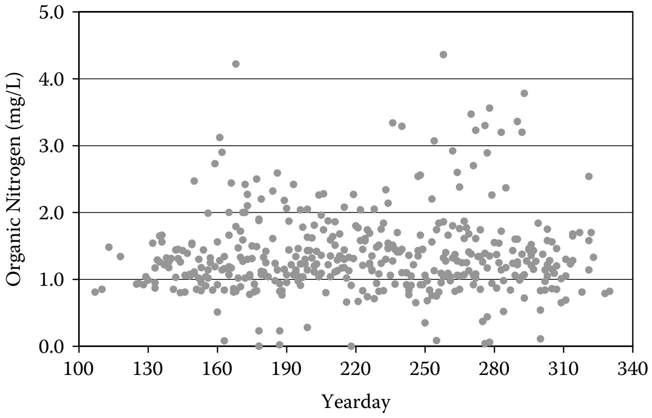


FIGURE 9.19 Organic nitrogen in the effluent from the Estevan, Saskatchewan, constructed wetland. Data are weekly during the unfrozen season 1994–2003, with an arithmetic mean of 1.58 mg/L. (Unpublished data from town of Estevan.)

Background Concentrations in FWS Wetlands

Because a portion of the background is due to decay processes in the wetland ecosystem, there is an effect of overall nutrient loading on the background. Lightly loaded wetlands that receive very little nitrogen or phosphorus possess lower backgrounds, such as the Orlando Easterly Wetland system in Florida (about 0.6 mg/L) or the Des Plaines, Illinois, wetlands (0.6–1.0 mg/L). Treatment wetlands that receive lagoon or secondary effluent are more heavily fertilized, and produce backgrounds of 1.5–2.0 mg/L.

There is not a large seasonality for background organic nitrogen. Wetlands operated at low hydraulic loadings have outlet concentrations approximating background. Examination of both northern and southern systems shows little seasonality, as typified by the Estevan, Saskatchewan, wetland, which operates during the unfrozen season (Figure 9.19).

Background Concentrations in HSSF Wetlands

Analysis of C_i versus C_o data for HSSF wetlands suggests there is a background concentration (C^*) in the range of 1–3 mg/L (Figure 9.20); a background concentration of 1.0 mg/L has been presumptively assumed for the rate constant analysis presented in this book.

However, it should be noted that several factors influence the organic nitrogen C^* range. Plant biomass cycling will return approximately 36 g/m²·yr of organic nitrogen back to the water column (accounted for in Figure 9.20). However, if the HSSF wetland bed is insulated with a mulch layer, the presence of this mulch material can exert an additional organic nitrogen loading on the system, especially if poorly decomposed mulch materials such as wood chips or tree bark are used. Data presented in Wallace *et al.* (2001) indicates that degradation of mulch materials can lead to TKN effluent concentrations in the range of 40–60 mg/L, and this elevation can continue for two to three years. Well-decomposed mulch materials such as peat or yard waste compost

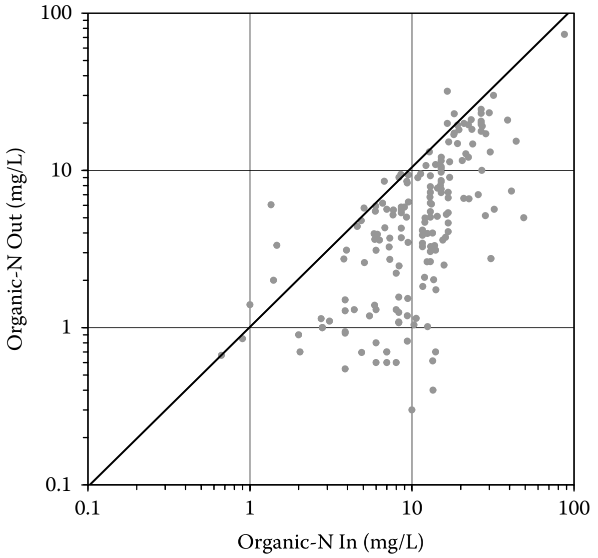


FIGURE 9.20 Outlet organic nitrogen as a function of inlet concentration for HSSF wetlands. Data are annual averages for 193 wetland-years from 116 wetland cells.

will return much lower effluent concentrations, in the range of 10–30 mg/L TKN.

Rates and Rate Constants

In conventional activated sludge treatment system design, ammonification is assumed to pertain to soluble organic nitrogen, and is modeled as a second-order process, first-order in soluble organic nitrogen and first-order in the biomass of heterotrophic microorganisms (U.S. EPA, 1993b). The ammonification rate increases with a doubling of the rate constant for a temperature increase of 10°C ($\theta = 1.07$) (U.S. EPA, 1993b).

The rate of ammonification in flooded soils also depends on temperature and pH (Reddy and Patrick, 1984). The ammonification rate increases with a doubling of the rate constant for a temperature increase of 10°C ($\theta = 1.07$). The rate of organic N mineralization was shown to increase with increasing temperature, from 5 to 35°C (Stanford *et al*., 1973). The θ-values are close to 1.07 in a temperature range of 15–35°C, but slightly higher ($\theta = 1.08$) at lower temperatures, 5–15°C. Mineralization essentially ceases when soil is frozen. The optimum pH range for ammonification is between 6.5 and 8.5 (Reddy and Patrick, 1984).

The organic nitrogen designation represents a large group of contributing forms and compounds. A large portion of organic nitrogen in wastewaters is likely to be particulate, although the particle size may be very small, resulting from bacterial debris and colloidal materials. A large part of the particulate organic fraction may be biodegradable (U.S. EPA, 1993b). The remaining portion comprises a potentially large number of soluble materials, ranging from the polypeptide components of humic substances to simple amino acids and urea (Fuchsman, 1980). Very few wetland studies have attempted to distinguish between dissolved and particulate forms. However, the Imperial, California, FWS project found that particulate organic nitrogen was not reduced through the train of wetland cells, whereas dissolved organic nitrogen was somewhat reduced, thus leaving a background of both particulate and dissolved forms.

Organic Nitrogen Rate Constants in FWS Wetlands

The loss of organic nitrogen in treatment wetland environments is here assumed to follow a first-order model, although there are but few studies that document the requisite decreasing profile through the wetland. For instance, the first-order assumption was made by Gerke *et al*. (2001) for particulate organic nitrogen removal in an FWS wetland. Such profiles were determined in the Listowel, Ontario, project, and displayed virtually no seasonality or temperature effect (Figure 9.21). The Listowel profiles show a decline to

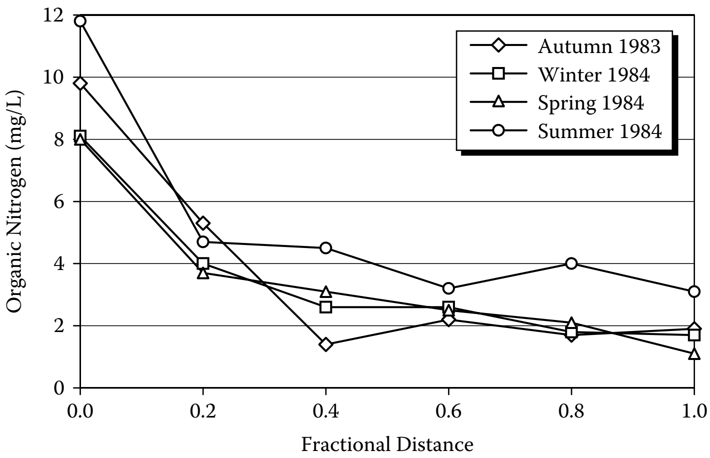


FIGURE 9.21 Organic nitrogen profiles through the Listowel, Ontario, FWS system 4 during all seasons. Samples were taken weekly, except biweekly in winter. The flow was collected in a culvert at each measurement point. (Data from Herskowitz (1986) *Listowel Artificial Marsh Project Report*. Ontario Ministry of the Environment, Water Resources Branch, Toronto, Ontario.)

a background plateau, which supports the concept of a background concentration.

Accordingly, an area-based first-order removal rate is utilized here:

$$J_{ON} = k_{ON}(C_{ON} - C^*_{ON}) \tag{9.39}$$

where
C_{ON} = wetland organic nitrogen concentration, mg/L
C^*_{ON} = background wetland organic nitrogen concentration, mg/L
J_{ON} = removal rate of organic nitrogen, g/m^2·yr
k_{ON} = removal rate constant for organic nitrogen, m/yr

The wetland environment may have actual hydraulics ranging from a few tanks in series (TIS) up to a large number, approximating plug flow, depending on design. However, organic nitrogen is expected to show weathering effects due to its complex speciation, thus reducing the effective number of TIS (see Chapter 6). Accordingly, the *P*-*k*-*C** model is chosen, with $P < N$. To compare results across systems that in general do not have known *N*-values, the value $P = 3$ is chosen here. Further, there is a fairly narrow band of *C** values, and therefore, $C^* = 1.5$ mg/L is chosen here to allow comparisons. The remaining model parameter is the *k*-value, selected to fit the model:

$$\frac{C_{ON,out} - C^*}{C_{ON,in} - C^*} = \left(1 + \frac{k_{ON}}{3q}\right)^{-3} \tag{9.40}$$

where
q = wetland hydraulic loading, m/yr
C^* = 1.5 mg/L (assumed)

Because of the selection of $C^* = 1.5$, parameter estimation is not reliable for low inlet concentrations, and those wetlands with $C_{ON,in} < 5$ mg/L have been excluded from calibration. Out of 147 wetlands with data for organic nitrogen (Figure 9.17), 60 systems met this criterion.

There appears to be little or no temperature dependence of organic nitrogen *k*-values. This concept is based upon intrasystem calibrations for individual wetlands. For example, Gerke *et al.* (2001) present data that indicate $\theta = 1.008$ for the Kingman, Arizona system. The Listowel systems calibrate to $\theta = 0.982$ for Equation 9.34, compared to 1.017 for the alternate assumption of plug flow (Kadlec and Reddy, 2001). This is in contrast to the strong temperature dependence observed in soils and mechanical activated sludge treatment systems.

Results of calibration of *k*-values for entire periods of record for the qualifying FWS wetland are summarized in Table 9.11. The median *k*-value for organic nitrogen is 17.3 m/yr, but the range is wide. The 10th–90th percentile range is 5.0–61.9 m/yr. Accordingly, there is a large design window that encompasses varying degrees of risk. Figure 9.17 may be used to place a proposed design hydraulic loading and inlet organic N concentration in the perspective of an existing database.

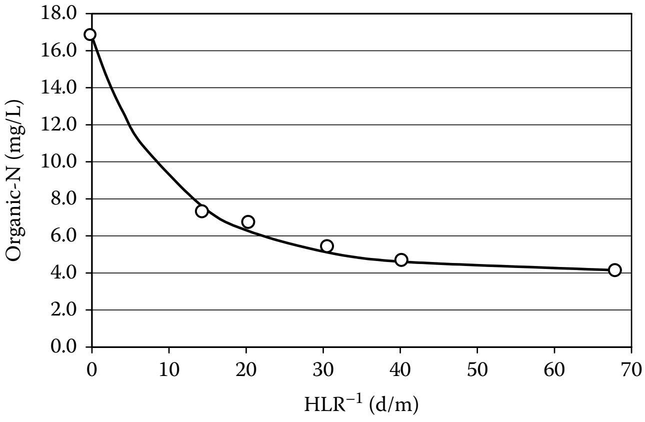


FIGURE 9.22 Decline of organic nitrogen with detention time (inverse HLR) for side-by-side HSSF wetlands receiving dairy effluent. The line is a *P*-*k*-*C** model with $P = 6$, $C^* = 4$ mg/L, and $k = 36$ m/yr ($R^2 = 0.96$). (Data from Tanner *et al.* (1998b) *Journal of Environmental Quality*, 27(2): 448–458.)

Organic Nitrogen Rate Constants in HSSF Wetlands

The *P*-*k*-*C** model can also be used to fit the reduction of organic nitrogen in HSSF wetlands (Figure 9.22), as the reduction appears to be first-order and decline to a nonzero background concentration (*C**).

The *P*-*k*-*C** model can be used to determine *k*-rates for organic nitrogen. Since organic nitrogen is a collection of individual nitrogenous compounds (including particulate matter) that undergo weathering in the wetland, the parameter *P* will always be less than the hydraulic parameter number of tanks in series (*N*TIS). Relatively few HSSF wetlands have been tracer tested; so the hydraulic parameter *N*TIS is not known with certainty. For a data set of 37 tracer-tested HSSF wetlands, the median value was *N*TIS = 11 (see Chapter 6). To account for weathering effects, *P*TIS = 6 has been assumed in the determination of annual rate constants. A background concentration $C^* = 1.0$ mg/L has also been assumed (see Figure 9.20).

Results of calibration for average annual *k*-rates are summarized in Table 9.12. The median *k*-value is 19.6 m/yr; but the range of *k*-values is wider than that observed in FWS wetlands. The 10th–90th percentile range is 3.8–124.2 m/yr. As a result, there is a wide range of *k*-rates that can be selected for design, with varying degrees of risk. Figure 9.19 can be used evaluate a particular design selection of *k* in the context of the existing performance database for HSSF wetlands.

There appears to be little temperature dependence or organic nitrogen *k*-values. Data from 12 HSSF wetlands yield a median value of $\theta = 1.009$, with a 10th–90th percentile range of 0.982–1.047, as indicated in Table 9.13.

TABLE 9.13
Temperature Coefficients for Ammonification Rate Constants in HSSF Wetlands

Site	Reference	Cell	T range (°C)	Mean HLR (cm/d)	Mean C_i (mg/L)	Mean C_o (mg/L)	Theta
Richmond, NSW	Bavor *et al.* (1988)	Gravel	11–24	3.8	8.29	1.08	1.009
Richmond, NSW	Bavor *et al.* (1988)	*Typha*	11–24	4.6	8.29	1.56	1.016
Richmond, NSW	Bavor *et al.* (1988)	*Schoenoplectus*	11–24	5.1	8.29	0.01	1.003
Richmond, NSW	Bavor *et al.* (1988)	Mixed A	11–24	4.6	8.29	1.08	0.982
Richmond, NSW	Bavor *et al.* (1988)	Mixed B	11–24	3.8	8.29	1.09	0.956
Grand Lake, Minnesota	Unpublished data	1	1–17	1.0	7.2	3.3	0.992
Lincoln, Nebraska	Vanier and Dahab (1997)	*Typha, Schoenoplectus*	4–21	9.5	11.5	5	0.982
Hamilton, New Zealand	Tanner *et al.* (1998b)	L2	10–25	2.5	26.71	5	1.057
Hamilton, New Zealand	Tanner *et al.* (1998b)	L3	10–25	3.3	26.71	5	1.017
Hamilton, New Zealand	Tanner *et al.* (1998b)	L4	10–25	4.9	26.71	7	1.030
Hamilton, New Zealand	Tanner *et al.* (1998b)	L5	10–25	6.9	26.71	9	1.047

Percentile	Theta
0.05	0.969
0.10	0.982
0.20	0.982
0.30	0.992
0.40	1.003
0.50	1.009
0.60	1.016
0.70	1.017
0.80	1.030
0.90	1.047
0.95	1.052

9.7 PERFORMANCE FOR TKN

The combination of ammonia and organic nitrogen, TKN, is subject to consideration as a group of compounds that are reduced in wetlands. This parameter is often regarded as representative of the total liability for ammonia nitrogen, and the presumed oxygen requirement for nitrification. Because TKN may contain a considerable proportion of ammonia, vegetation is involved in the consumption of TKN. The organic nitrogen component of TKN is added back to the water from the ecosystem decomposition processes; hence, there are important interactions with the plants (including algae) in the wetland. TKN is rarely, if ever, a regulated water quality parameter.

Loading Considerations

Since TKN measures both organic and ammonia nitrogen, interconversions between these two species is not a concern, provided that plant uptake is accounted for (for the ammonia component). Performance data can be represented by loading analysis and the *P*-*k*-*C** model.

Reduction of TKN in FWS Wetlands

The median net period-of-record removal rate for 101 FWS systems receiving more than 5 mg/L of TKN is 207 g/m^2·yr (Table 9.14). There is, however, wide variability among systems.

It is again useful to represent annual wetland performance as the effluent concentration produced (C_o) by a given LRI (= HLR × C_i) and concentration (C_i). In the broad context, multiple data sets are represented by a trend that shows increasing C_o with increasing LRI, with different groupings associated with each inlet concentration (Figure 9.23). The overall slope of the intersystem data on the log–log coordinates varies from near zero for low inlet concentrations to about 1.0 for high inlet concentrations. As for organic nitrogen, inlet loading is an insufficient design specification because hydraulic load and inlet concentration are not interchangeable factors in the load representation.

Reduction of TKN in HSSF Wetlands

The median annual-average removal rate for 123 HSSF wetlands (197 system-years of data) is 228 g/m^2·yr, as indicated in Table 9.15.

It is also useful to evaluate wetland performance (C_o) as a function of the inlet loading (Figure 9.23). Figure 9.24 represents data from 112 HSSF wetlands (198 system-years). In general, there is an overall upward trend of the outlet TKN concentration (C_o) in response to the inlet TKN loading, with a log–log slope of slightly less than 1.0. However, this apparent slope is in large measure due to the shift in inlet concentrations. When a particular inlet concentration group (like those shown on Figure 9.24) is considered, the change in outlet TKN concentration is much less, as the intersystem slope for each

TABLE 9.14
Annual Reduction TKN in FWS Wetlands

Stipulations

1. Data restricted to wetlands receiving inlet $C > 5$ mg/L TKN.
2. Period of record averages are used in calculations.
3. For *k*-value calculations, the following *P*-*k*-*C** parameters are selected:
   a. $C^* = 1.5$ mg/L
   b. $P = 3$ TIS
4. Ranges of variables:

	HLR (cm/d)	TKN In (mg/L)	TKN Out (mg/L)
Mean	7.2	101.8	59.9
Median	3.9	32.4	20.0
Max	110.0	416.3	283.8
Min	0.6	5.1	1.7

Results (N = 101 wetlands)

Percentile	Load Removed (g/m²·yr)	Rate Coefficient (m/yr)
0.0	6	0.2
0.1	42	4.1
0.2	65	5.2
0.3	92	6.1
0.4	130	8.5
0.5	207	9.8
0.6	300	11.3
0.7	508	13.6
0.8	1,115	20.0
0.9	2,203	35.0
1.0	4,683	153.6

concentration grouping is approximately 0.3. This has important design implications, because as the hydraulic loading to the wetland is decreased, the reduction in effluent concentration follows the slope of the inlet concentration group, not the overall data set. Use of the overall data set will overpredict reduction in effluent TKN concentrations as the hydraulic load is decreased.

Reduction of TKN in VF Wetlands

Many vertical flow wetlands are designed with the express purpose of oxidizing organic and ammonia nitrogen. Effluent concentrations for TKN for vertical flow systems are summarized in Figure 9.25, which summarizes the period of record for 20 VF wetlands, annual averages for another 6 VF wetlands (17 system-years of data), plus data from intermittent sand filters that operate under similar loading and unsaturated flow conditions as VF wetlands (17 system-years of data). As Figure 9.25 illustrates, TKN loading is not an effective predictor of effluent TKN concentrations.

Background Concentrations of TKN

Treatment wetlands data display decreases in TKN with contact time, which are consistent with first-order reduction kinetics; but show a nonzero background concentration for long detention. This is consistent with the observed small background concentrations of organic N. As shall be discussed in this chapter, there is a zero background for ammonia, so background TKN is the same as background organic nitrogen, typically in the range of 0.5–2.0 mg/L for both FWS and HSSF wetland systems. For rate analysis, a background concentration (C^*) value of 1.5 mg/L was assumed for FWS wetlands (Table 9.14), and a value of 1.0 mg/L was assumed for HSSF wetlands (Table 9.15).

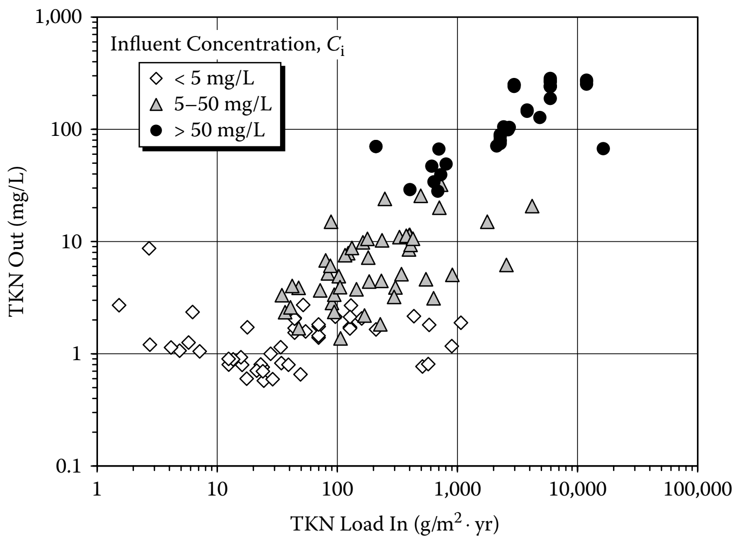


FIGURE 9.23 Load–concentration plot for total Kjeldahl nitrogen in FWS wetlands. Points are separated according to the inlet concentration range. Each point represents the period of record (POR) for one of 135 wetlands.

TABLE 9.15
Annual Reduction of Total Kjeldahl Nitrogen in HSSF Wetlands

Stipulations

1. The decomposition of 2,000 g/m²·yr of biomass causes production of 36 gN/m²·yr of organic nitrogen.
2. Annual averages are used in calculations.
3. For *k*-value calculations, the following *P*-*k*-*C** parameters are selected:
   a. $C^* = 1.0$ mg/L
   b. $P = 6$ TIS
4. Ranges of variables:

	HLR (cm/d)	TKN In (mg/L)	TKN Out (mg/L)
Mean	7.6	49.3	32.5
Median	4.9	34.8	23.2
Max	41.2	226.0	189.5
Min	1.1	2.1	0.4

Results (N = 123; N·*t* = 197 wetland-years)

Percentile	Load Removed (g/m²·yr)	Rate Coefficient (m/yr)
0.05	−180	−18.5
0.10	41	1.8
0.20	109	3.4
0.30	151	4.8
0.40	185	7.1
0.50	228	9.1
0.60	287	11.6
0.70	361	14.6
0.80	453	19.0
0.90	585	37.5
0.95	1,761	144.2

Rates and Rate Constants

In conventional activated sludge treatment system design, removal of TKN is not directly modeled, but results from ammonification of the organic component and nitrification of the ammonia component. The loss of organic nitrogen in treatment wetland environments is here assumed to follow a first-order model, based upon studies that document the requisite decreasing profile through the wetland.

Profiles along the length of the Kingman, Arizona, FWS system show such decreases, but removal is different in warm and cold seasons (Figure 9.26). Accordingly, an area-based first-order removal rate is utilized here:

$$J_{TKN} = k_{TKN}(C_{TKN} - C^*_{TKN}) \tag{9.41}$$

where

C_{TKN} = wetland TKN concentration, mg/L
C^*_{TKN} = background wetland TKN concentration, mg/L
J_{TKN} = removal rate of TKN, g/m²·yr
k_{TKN} = removal rate constant for TKN, m/yr

The wetland environment may have actual hydraulics ranging from a few TIS up to a large number, approximating plug flow, depending on wetland configuration. Organic nitrogen is expected to show weathering effects as discussed above. Ammonia is less liable to experience weathering, because it exists primarily in dissolved form, typically with only small contributions of particulate (sorbed) forms. Speculatively, the effective number of TIS (see Chapter 6) should be less than the tracer TIS, but by a slightly lesser margin than for

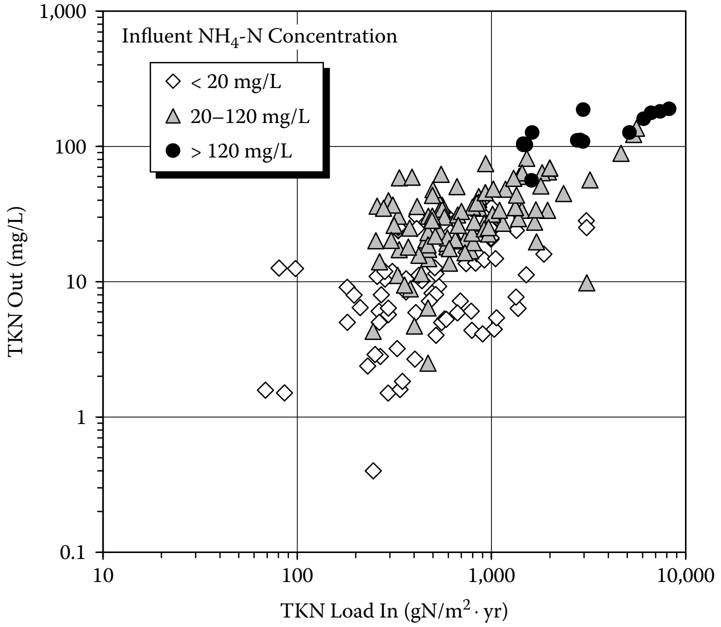


FIGURE 9.24 Load response data for total Kjeldahl nitrogen in HSSF wetlands. Annual average information from 112 wetlands and 198 wetland-years is shown.

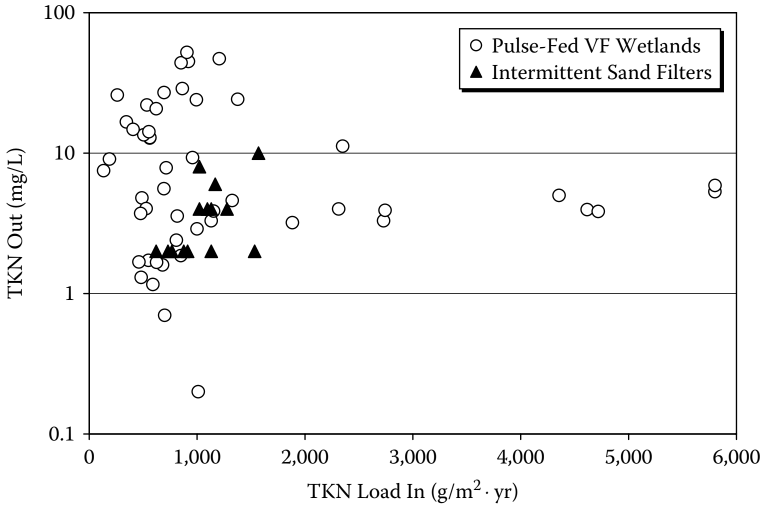


FIGURE 9.25 Concentration–loading chart for TKN in pulse-fed vertical flow wetlands and intermittent sand filters. Data includes period-of-record performance for 20 vertical flow wetlands, annual average reductions for another 6 vertical flow wetlands (17 system-years of data), and annual average reductions for three intermittent sand filters (17 system-years of data) that were operated under similar loading regimes. TKN loading is not an effective predictor of effluent TKN concentrations.

organic nitrogen. Accordingly, the *P-k-C** model is chosen, with $P < N$.

TKN Rate Constants for FWS Wetlands

To compare results across systems that in general do not have known *N*-values, the value $P = 3$ is chosen here. The value $C^* = 1.5$ mg/L is retained based upon organic nitrogen considerations. The remaining model parameter is the *k*-value, selected to fit the model:

$$\frac{C_{\mathrm{TKN,out}} - 1.5}{C_{\mathrm{TKN,in}} - 1.5} = \left(1 + \frac{k_{\mathrm{TKN}}}{3q}\right)^{-3} \tag{9.42}$$

Because of the selection of $C^* = 1.5$, parameter estimation is not reliable for low inlet concentrations, and wetlands with $C_{\mathrm{TKN,in}} < 5$ mg/L have been excluded from calibration. Out of 157 wetlands with data for TKN (Figure 9.23), 101 met this criterion. The median annual rate constant was $k_{\mathrm{TKN}} = 9.8$ m/yr (Table 9.14). The 10th–90th percentile range is 4.1–35.0 m/yr. There is a significant temperature dependence of TKN *k*-values. Even on an average annual basis, temperature or season may be an important determinant of the rate constant, and is thus responsible for the some of the intersystem variability in annual *k*-values. Accordingly, it is necessary to examine intra-annual effects.

Microbially Dominated Wetlands

When the TKN loading to the wetland exceeds the growth requirements of the plants and algae by a considerable margin, the removal of TKN is very likely to be microbially mediated. The loading limit for bacterial conversion to predominate is approximately 120 gN/m^2·yr (Kadlec, 2005d).

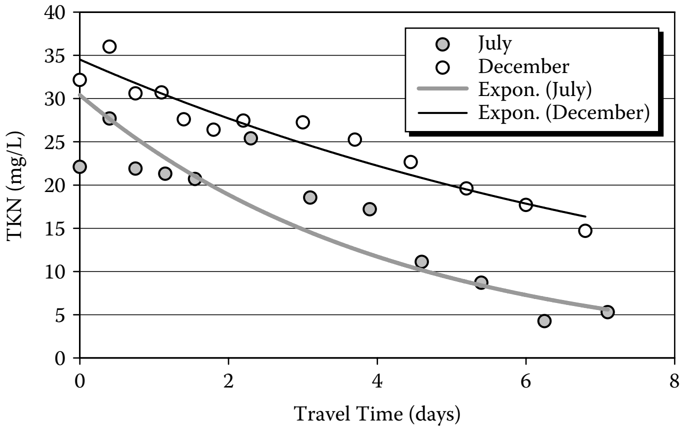


FIGURE 9.26 Longitudinal profiles of TKN at the Kingman, Arizona, FWS wetland. (Data from Gerke *et al.* (2001) *Water Research*, 35(16): 3857–3866.)

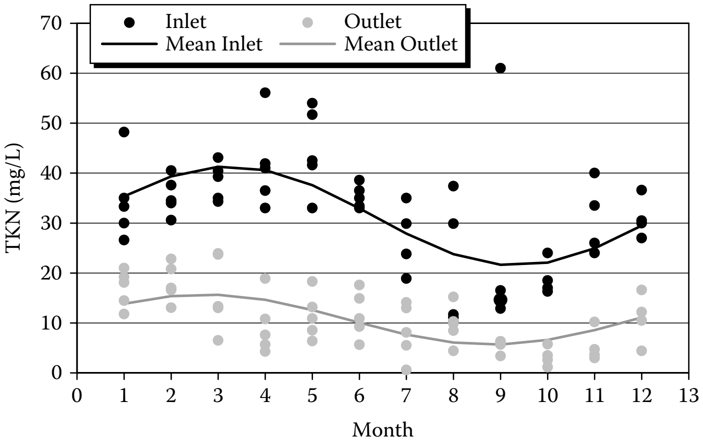


FIGURE 9.27 Folded inlet and outlet time series for TKN for the Kingman, Arizona, FWS wetland. (Unpublished data from city of Kingman.)

Cyclic parameters	Inlet	Outlet
C_{mean} (mg/L)	31.4	10.7
A	0.32	0.47
t_{max} (days)	83	66
t_{min} (days)	265	249

There is typically a monotonic decline in TKN along the flow path of a wetland (see Figure 9.26). Sampling along the flow direction results in variability from at least two sources: (1) spatial selection of the sampling points, and (2) temporal variability in input flows and concentrations that may propagate in the flow direction. Nevertheless, there is a clear downward trend, as TKN is removed from the water during travel through the wetland. Rates of decline are faster in summer than in winter, implying that a temperature effect is present in these microbially dominated systems. In many wetland systems, there are annual trends in input concentrations that often follow a sinusoidal tend, reflecting changes in the pretreatment and inlet water quality for that pretreatment wetland (Figure 9.27). Under these circumstances, it is not appropriate to use percentage reductions as a measure of performance, because of the confounding effects of seasonal flows, concentrations, and microbial activity. Accordingly, the first-order model is here utilized, together with a temperature coefficient (θ), which are capable of accounting for these effects (see Chapter 6).

Results of calibration of k-values for entire periods of record for representative wetlands are summarized in Table 9.16. Monthly averages were used to avoid synoptic error (transit time offset). Calibrations were performed for best estimates of the internal hydraulics for each wetland. Therefore, P-values range from 2 (New Hanover, measured $P = N = 2$) to near plug flow conditions, based upon system geometry. In most cases, the $C^* = 1.5$ was used, excepting three cases in which slightly different C^* were indicated by data. The median k_{20}-value for TKN is 21.0 m/yr, but the range is wide.

Temperature coefficients had a median value of 1.036, indicating a relatively strong thermal effect on the suite of microbial processes that contribute to TKN reduction.

The example systems in Table 9.16 do not display any limitations due to the supplies of oxygen. The theoretical oxygen demands for full nitrification of the removed TKN are in the range of 0–7.1 g/m^2·d, which is within the feasible range of reaeration combined with inlet dissolved oxygen. There was generally some BOD entering these example systems, with a median of 1.5 times the entering TKN. This potential carbonaceous oxygen demand does not contribute to an extreme need for DO in the example systems, although it may contribute to less than optimal nitrification. The role of open water in providing the oxygen for nitrification is not clear in this intersystem comparison of rate constants for TKN, because of confusion with other factors.

Agronomic Wetlands (Lightly Loaded Systems)

When the TKN loading to the wetland is less than the growth requirements of the plants and algae by a considerable margin, the removal of TKN is very likely to be mediated by the growth and decay of biomass. As a rough guideline, this situation occurs for TKN loading less than approximately 120 gN/m^2·yr (Kadlec, 2005d). This occurs for almost half (41%) of the 135 wetlands displayed in Figure 9.23. It is important to note that low inlet TKN load very often means very low inlet TKN concentration, close to background; consequently, there is no ability to obtain meaningful calibrations of TKN rate constants.

Uptake presumably occurs for the ammonia component of TKN, and release may be considered to add to the organic component. Because plant uptake rates do not correspond to the annual cycle of water temperatures, TKN removal in agronomic wetlands cannot be characterized by modified Arrhenius θ-factors. For example, the Estevan, Saskatchewan, system had modest hydraulic loadings coupled with low

TABLE 9.16
Dependence of TKN Rate Constants on Temperature for FWS Systems

Site	Location	TKN In (mg/L)	TKN Out (mg/L)	TKN Load ($g/m^2 \cdot yr$)	*P* (TIS)	*C** (mg/L)	k_{20} (m/yr)	θ	Estimated Open Water (%)	DO (mg/L)	Annual T (°C)	TKN Theoretical O_2 Demand ($g/m^2 \cdot d$)	BOD/TKN In
Texel	The Netherlands	3.2	1.8	36–1,279	∞	1.8	118	1.075	50	4.5–16	10.0	0.2–7.1	—
Imperial	California	3.3	2.1	368	∞	1.25	81	1.036	77	7.5–10.1	20.7	1.7	1.9
Brawley	California	4.7	1.8	206	∞	1.25	70	1.029	75	8.2–10.9	20.7	1.6	1.4
Eskilstuna	Sweden	4.9	3.6	205	6	1.5	21	1.011	80	—	6.5	0.7	1.3
Brighton	Ontario	9.9	11.8	172	3	1.5	–4.4	1.045	20	—	9.4	–0.4	0.5
Listowel 4	Ontario	18.5	8.8	132	10	1.5	4	0.969	5	0.5–5	8.0	0.9	3.0
Listowel 5	Ontario	18.5	10.6	178	3	1.5	8	1.011	5	0.5–3.3	7.9	1.0	3.0
Linköping	Sweden	18.6	10.8	288	3	1.5	17	1.037	10	—	6.7	1.5	—
Kingman	Arizona	33.0	11.0	501	20	1.5	25	1.022	4	2.8–7.3	11.7	4.2	1.5
Warangal	India	36.5	4.0	570	3	1.5	66	1.081	—	4	23.1	6.4	4.4
New Hanover	North Carolina	132.0	66.0	716	2	1.5	4.5	1.078	10	3.3–4.2	19.4	4.5	0.4
Median						**1.5**	**21.0**	**1.036**			**10.0**	**1.56**	**1.47**

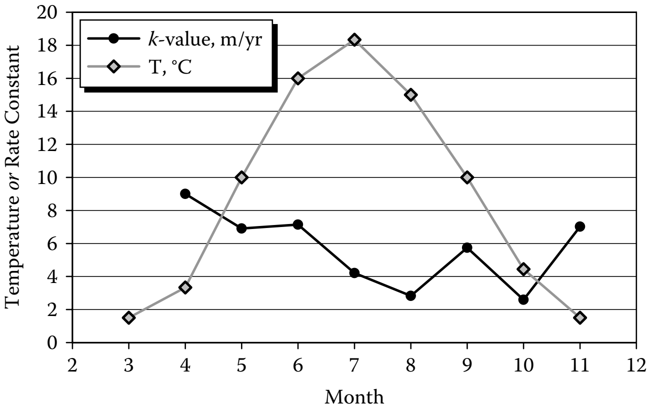


FIGURE 9.28 The absence of a relationship between temperature and total Kjeldahl nitrogen rate constants for the Estevan, Saskatchewan, FWS system. Nine years' data are represented, for which the TKN loading was 47 g/m²·yr, well below the agronomic rate of 120 g/m²·yr.

incoming TKN from lagoon pretreatment. This system was operated seasonally (May–October) for nine years, during which time the TKN loading averaged 47 g/m²·yr. Although estimated temperatures ranged from less than 2°C to about 20°C, there was no corresponding change in monthly *k*-values for TKN (Figure 9.28).

For agronomic wetlands, there may still be seasonality, out of synchronization with temperature. The appropriate method of dealing with this situation is via monthly *k*-values, according to (see Chapter 6):

$$k = k_j \tag{9.43}$$

where
j = month number = 1, 2, 3, ... 12

For the nine-year period of record for Estevan, the mean monthly results are:

$$\begin{array}{llllllllll} k_{\mathrm{TKN},j} & = 9.0 & 6.9 & 7.1 & 4.2 & 2.8 & 5.7 & 2.6 & 7.0\ \mathrm{m/yr} \\ j & = 4 & 5 & 6 & 7 & 8 & 9 & 10 & 11 \end{array} \tag{9.44}$$

In any case, it is prudent to examine the actual seasonal progression of *k*-values, to ascertain whether a θ-value approach is warranted or not. Additionally, if there is not a temperature correlation, the R^2 for a θ-factor regression will be very low.

TKN Rate Constants for HSSF Wetlands

Ranges in annual average *k*-rates for HSSF wetlands are summarized in Table 9.15, with the assumptions of PTIS = 6; C^* = 1.0 mg/L. The median *k*-rate is 9.1 m/yr. However, there is wide variability in the data, and the 10th–90th percentile range is 1.8–37.5 m/yr.

There appears to be little, if any, temperature dependence on TKN removal in HSSF wetlands. Data from 9 HSSF wetlands are presented in Table 9.17. The median θ-factor was 1.001; and the 10th–90th percentile range is 0.951–1.011.

TABLE 9.17
Temperature Coefficients for TKN Removal Rate Constants in HSSF Wetlands

Site	Reference	Cell	T range (°C)	Mean HLR (cm/d)	Mean C_i (mg/L)	Mean C_o (mg/L)	Theta
Richmond, NSW	Bavor *et al.* (1988)	Gravel	11–24	3.8	43.5	19.9	1.001
Richmond, NSW	Bavor *et al.* (1988)	*Typha*	11–24	4.6	43.5	20.4	1.001
Richmond, NSW	Bavor *et al.* (1988)	*Schoenoplectus*	11–24	5.1	43.5	19.4	1.006
Richmond, NSW	Bavor *et al.* (1988)	Mixed A	11–24	4.6	43.5	16.4	1.005
Richmond, NSW	Bavor *et al.* (1988)	Mixed B	11–24	3.8	43.5	13.7	0.998
Benton, Kentucky	TVA unpublished data	3	5–25	8.4	14.3	10.2	0.954
Grand Lake, Minnesota	Unpublished data	1	1–17	1.0	58.4	41.2	1.008
Lincoln, Nebraska	Vanier and Dahab (1997)	*Typha, Schoenoplectus*	4–21	9.5	31.8	21.0	1.024
Waipoua, New Zealand	Unpublished data	1	11–21	0.4	74.9	61.7	0.940
Richmond, NSW	Bavor *et al.* (1988)	Gravel	11–24	3.8	43.5	19.9	1.001
Richmond, NSW	Bavor *et al.* (1988)	*Typha*	11–24	4.6	43.5	20.4	1.001

Percentile	Theta
0.10	0.951
0.20	0.980
0.30	0.999
0.40	1.001
0.50	1.001
0.60	1.004
0.70	1.006
0.80	1.007
0.90	1.011

9.8 PERFORMANCE FOR TOTAL NITROGEN

TN, which is defined as the combination of organic, ammonia, and oxidized nitrogen, is subject to consideration as a group of compounds that are reduced in wetlands. This grouping is known to possess sequential conversions, including primarily ammonification followed by nitrification, followed by denitrification, all proceeding at varying rates. TN in the water is augmented by releases from decaying vegetation and microbial biomass. As a consequence, the rate of decline of TN along the flow path is expected to be determined in part by the speciation of the incoming nitrogen. TN removal rates for nitrified influents are anticipated to be highest, because the precursor conversions of organic and ammonia nitrogen have already occurred in pretreatment.

Loading Considerations

Reductions in TN in treatment wetlands systems can be represented by loading analysis and the *P-k-C** model.

Reduction of Total Nitrogen in FWS Wetlands

The median net period-of-record removal rate for 116 FWS systems receiving more than 5 mg/L TN is 129 g/m^2·yr (Table 9.18). There is, however, wide variability among systems.

It is again useful to represent annual wetland performance as the effluent concentration produced (C_o) by a given inlet loading rate (LRI = HLR × C_i) and concentration (C_i). In the broad context, multiple data sets are represented by a trend that shows increasing C_o with increasing LRI, with different groupings associated with each inlet concentration (Figure 9.29). The overall slope of the intersystem data on the log–log coordinates varies from near zero for low inlet concentrations to about 1.0 for high inlet concentrations.

TABLE 9.18
Annual Reduction of Total Nitrogen in FWS Wetlands

Stipulations

1. Data restricted to wetlands receiving inlet $C > 5$ mg/L TN.
2. Period of record averages are used in calculations.
3. For *k*-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 1.5$ mg/L
   b. $P = 3$ TIS
4. Ranges of variables:

	HLR (cm/d)	TN In (mg/L)	TN Out (mg/L)
Mean	8.9	83.8	49.5
Median	4.3	17.4	10.2
Max	123.0	416.6	284.6
Min	0.2	5.2	1.7

Results (N = 116 wetlands)

Percentile	Load Removed (g/m^2·yr)	Rate Coefficient (m/yr)
0.0	3	0.2
0.1	42	4.0
0.2	67	5.3
0.3	76	6.6
0.4	103	8.7
0.5	129	12.6
0.6	214	17.1
0.7	375	24.2
0.8	550	29.6
0.9	1,973	39.2
1.0	7,504	109.0

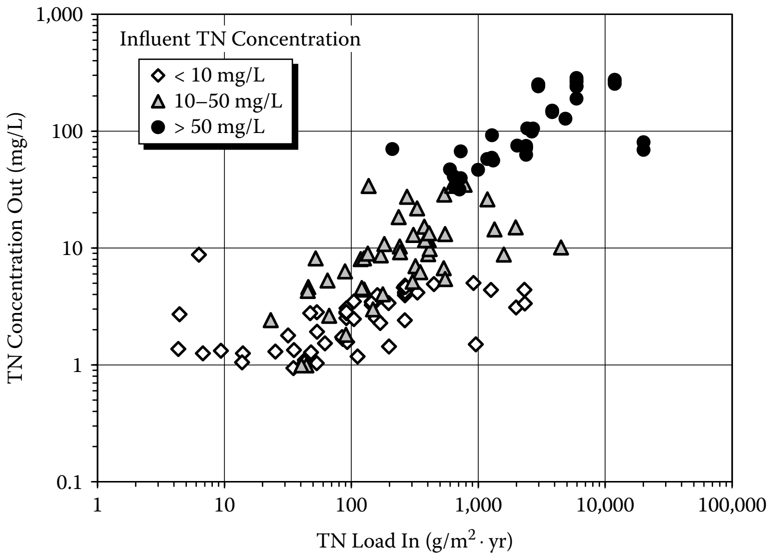


FIGURE 9.29 Load–concentration plot for total nitrogen in FWS wetlands. Points are separated according to the inlet concentration range. Each point represents the entire period of record (POR) for one of 141 wetlands.

As for TKN and organic nitrogen, inlet loading is an insufficient design specification, because hydraulic load and inlet concentration are not interchangeable factors in the load representation.

Reduction of Total Nitrogen in HSSF Wetlands

The median annual-average removal rate for 123 HSSF wetlands (197 system-years of data) is 273 g/m^2·yr, as indicated in Table 9.19.

It is also useful to evaluate wetland performance (C_o) as a function of the inlet loading (Figure 9.30). Figure 9.30 represents data from 112 HSSF wetlands (198 system-years). In general, there is an overall upward trend of the outlet TN concentration (C_o) in response to the inlet TN loading, with a log–log slope of slightly less than 1.0. However, this apparent slope is in large measure due to the shift in inlet concentrations. When a particular inlet concentration group (like those shown on Figure 9.30) is considered, the change in outlet TN concentration is much less, as the intersystem slope for each concentration grouping is approximately 0.3. This has important design implications, because as the hydraulic loading to the wetland is decreased, the reduction in effluent concentration follows the slope of the inlet concentration group, not the overall data set. Use of the overall data set will overpredict the reductions in effluent TN concentrations as the hydraulic loading is decreased.

Reduction of Total Nitrogen in VF Wetlands

Vertical flow wetlands typically transform organic and ammonia nitrogen to oxidized forms (nitrate and nitrite). As a result, effluents from VF wetlands are typically dominated by oxidized forms of nitrogen and the overall reduction of TN may be low (although the chemical form of nitrogen exiting the wetland may be very different than the chemical form entering the wetland).

TABLE 9.19
Annual Reduction of Total Nitrogen in HSSF Wetlands

Stipulations

1. The decomposition of 2,000 g/m^2·yr of biomass causes production of 36 gN/m^2·yr of organic nitrogen.
2. Annual averages are used in calculations.
3. For *k*-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 1.0$ mg/L
   b. $P = 6$ TIS
4. Ranges of variables:

	HLR (cm/d)	TN In (mg/L)	TN Out (mg/L)
Mean	7.6	54.1	36.0
Median	4.9	41.4	25.6
Max	41.2	250.6	190.6
Min	1.1	6.8	3.4

Results (N = 123; N·*t* = 198 wetland-years)

Percentile	Load Removed (g/m^2·yr)	Rate Coefficient (m/yr)
0.05	−135.9	−2.2
0.10	61.6	1.9
0.20	120.9	3.3
0.30	161.1	4.7
0.40	209.0	6.8
0.50	272.8	8.4
0.60	371.7	11.2
0.70	443.8	14.2
0.80	561.7	18.1
0.90	713.2	30.5
0.95	2,680.2	100.3

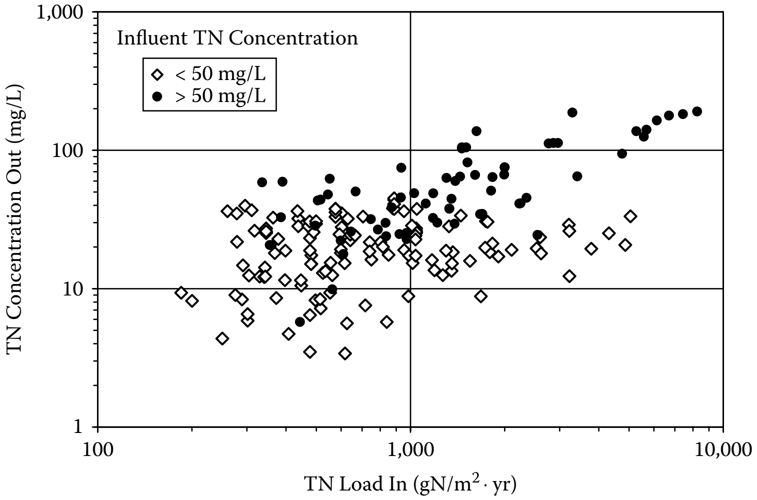


FIGURE 9.30 Load response data for total nitrogen HSSF wetlands. Annual average information from 112 wetlands and 198 system-years is shown.

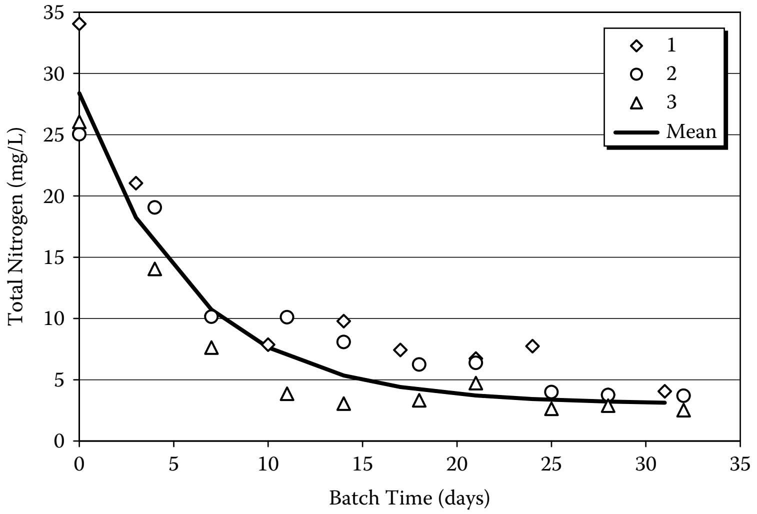


FIGURE 9.31 Reduction of total nitrogen in the batch treatment FWS wetland systems at Humboldt, Saskatchewan. (Data from Lakhsman (1981) *A demonstration project at Humboldt to provide tertiary treatment to the municipal effluent using aquatic plants.* SRC Publication No. E-820-4-E-81: Saskatchewan Research Council. Graph from Kadlec and Knight (1996). *Treatment Wetlands.* First Edition, CRC Press, Boca Raton, Florida.)

Background Concentrations of Total Nitrogen

Treatment wetlands data display decreases in TN with contact time, which are consistent with first-order reduction kinetics; but show a nonzero background concentration for long detention. This is true for the time series of TN values when the system is operated in batch mode (Figure 9.31), and is also true for side-by-side systems of different detention times operated in flow through mode (Figure 9.32).

These nonzero plateaus of TN concentration at long detention times are consistent with the observed small background concentrations of organic nitrogen, and reflected in TKN. As shall be discussed in the following text, there are zero background concentrations for ammonia and oxidized nitrogen, so background TN is essentially the same as background organic nitrogen, typically in the range of 0.5–2.5 mg/L for all types of treatment wetlands.

Rates and Rate Constants

In conventional activated sludge treatment system design, removal of TN is not directly modeled, but results from ammonification of the organic component, nitrification of the ammonia component, and denitrification of the oxidized nitrogen component. It is presumed in this section that TN removal in wetland treatment systems is an integrative measure of individual nitrogen transformations which can be approximated by area-based first-order rate expressions. This is supported by data such as that in Figures 9.31 and 9.32. Therefore, an area-based first-order removal rate is utilized here:

$$J_{TN} = k_{TN}(C_{TN} - C_{TN}^{*}) \tag{9.45}$$

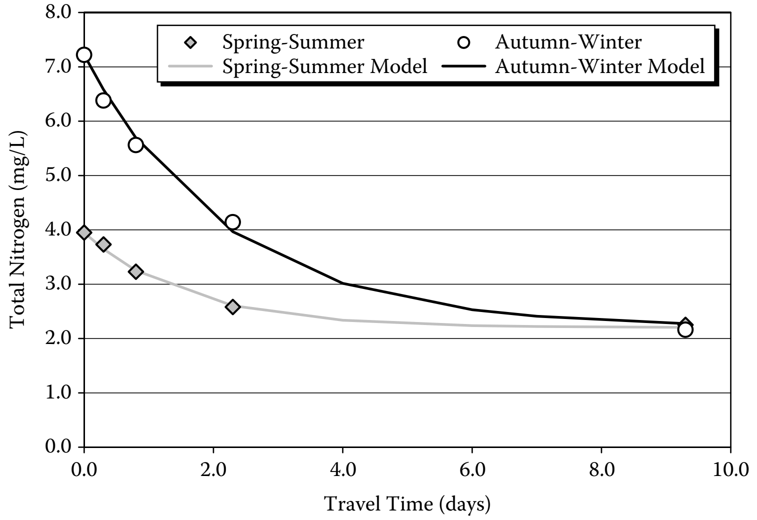


FIGURE 9.32 Total nitrogen as a function of nominal detention time for the Texel wetlands. Replicate channels were operated side-by-side with different flow rates. (Data from Toet (2003) A Treatment Wetland Used for Polishing Tertiary Effluent from a Sewage Treatment Plant: Performances and Processes. Ph.D. Thesis, University of Utrecht, The Netherlands.)

where

C_{TN} = wetland total nitrogen concentration, mg/L
C^*_{TN} = background wetland total nitrogen concentration, mg/L
J_{TN} = removal rate of total nitrogen, g/m²·yr
k_{TN} = removal rate constant for total nitrogen, m/yr

The wetland environment may have actual hydraulics ranging from a few TIS up to a large number, approximating plug flow, depending on wetland configuration. Organic nitrogen is expected to show weathering effects as discussed in the previous text. Ammonia and nitrate are less liable to experience weathering, because these exist primarily in dissolved form, typically with only small contributions of particulate (sorbed) forms. Speculatively, the effective number of TIS (see Chapter 6) will be less than the tracer TIS, but by a slightly lesser margin than for organic nitrogen and TKN. Accordingly, the P-k-C^* model is chosen, with $P < N$.

Total Nitrogen Rate Constants for FWS Wetlands

Results across systems for the assumed value $P = 3$ are summarized in this section. The value $C^* = 1.5$ mg/L is retained based upon organic nitrogen considerations. The remaining model parameter is the k-value, selected to fit the model:

$$\frac{C_{TN,\,out} - 1.5}{C_{TN,\,in} - 1.5} = \left(1 + \frac{k_{TN}}{3q}\right)^{-3} \qquad (9.46)$$

Because of the selection of $C^* = 1.5$, parameter estimation is not reliable for low inlet concentrations, and those wetland with $C_{TN,in} < 5$ mg/L have been excluded from calibration. Out of 141 wetlands with data for TN (Figure 9.29), 116 met this criterion. The median annual rate constant was k_{TN} = 12.6 m/yr (Table 9.18). The 10th–90th percentile range is 4.0–39.2 m/yr. There is a significant temperature dependence of TN k-values. Even on an average annual basis, temperature or season may be an important determinant of the rate constant, and is thus responsible for the some of the intersystem variability in annual k-values. Accordingly, it is necessary to examine intra-annual effects.

Microbially Dominated Wetlands

There are typically clear downward trends in concentration along the flow path, as TN is removed from the water during travel through the wetland. Summer rates of decline are greater than in winter, implying that a temperature effect is present in these microbially dominated systems. It is not appropriate to use percentage reductions as a measure of seasonal performance, because of the confounding effects of seasonal flows, concentrations, and microbial activity. In many wetland systems, there are annual trends in input concentrations that often follow a sinusoidal tend, reflecting changes in the pretreatment and inlet water quality for that pretreatment wetland (Figure 9.33). For instance, a pretreatment plant conducting partial nitrification and partial denitrification will be more effective in summer, leading to less ammonia and less nitrate in the warmer months. A temperature coefficient (θ), is capable of accounting for these effects (see Chapter 6).

Results of calibration of k-values for entire periods of record for representative wetlands are summarized in Table 9.20. Monthly averages were used to avoid synoptic

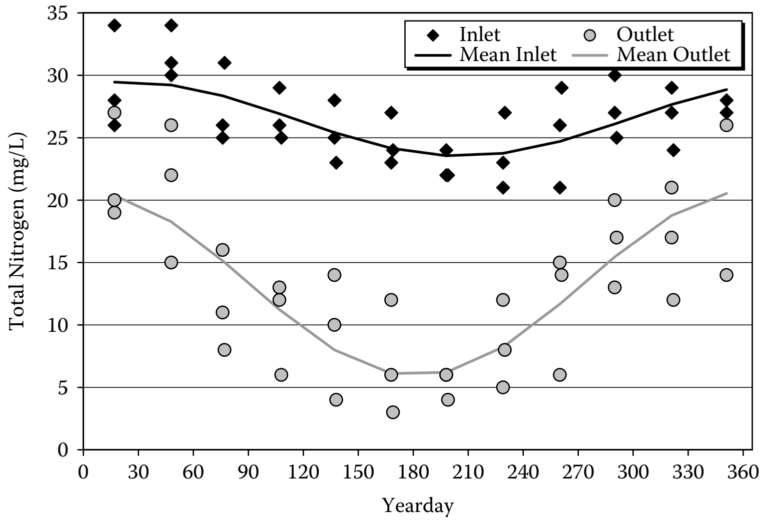


FIGURE 9.33 Folded time series of inlet and outlet total nitrogen for the Linköping, Sweden, FWS wetlands. (From unpublished data, courtesy of K. Tonderski.)

Cyclic parameters	Inlet	Outlet
C_{mean} (mg/L)	26.5	13.3
A	0.11	0.56
t_{max} (days)	24	364
t_{min} (days)	207	182

TABLE 9.20
Dependence of Total Nitrogen Rate Constants on Temperature for FWS Systems

Site	Location	System	Years	Estimated Open Water (%)	TN In (mg/L)	TN Out (mg/L)	TN Load (g/m²·yr)	DO (mg/L)	Annual (°C)	*P* (TIS)	*C** (mg/L)	k_{20} (m/yr)	Theta
Texel	The Netherlands	Pilot	1	50	5.6	2.2	63–2,242	4.5–16	10.0	∞	2.2	115	1.050
Tres Rios	Arizona	H1 pre	2	25	5.6	2.6	268	1.3–3.1	18.9	6	0.8	49	1.061
Tres Rios	Arizona	H2 pre	2	25	5.6	2.4	264	1.4–3.1	18.9	6	0.8	43	1.084
Tres Rios	Arizona	H1 post	2	85	6.5	4.4	270	3	19.9	6	0.8	13	1.130
Tres Rios	Arizona	H2 post	2	90	6.5	4.8	308	3.0–4.2	20.7	6	0.8	11	1.103
Imperial	California	All	4	77	6.6	3.8	709	7.5–10.1	20.7	∞	1.25	76	1.048
Brawley	California	All	4	75	7.6	2.3	257	8.2–10.9	20.7	∞	1.25	52	1.080
Lakeland	Florida	Cell 1	7	5	11.0	4.0	172	2.5–7.3	22.9	3	1.0	18	1.079
Brighton	Ontario	All	4	20	13.0	10.0	219	—	9.4	3	1.5	7	1.047
Listowel	Ontario	4	4	5	19.0	9.0	135	0.5–5	8.0	10	1.5	4	0.978
Listowel	Ontario	5	4	5	19.0	11.0	173	0.5–3.3	7.9	3	1.5	8	1.012
Eskilstuna	Sweden	Ekeby	3	80	20.0	15.0	1,175	—	6.5	6	1.5	25	1.049
Linköping	Sweden	All	3	10	27.0	13.0	410	—	6.7	3	1.5	42	1.081
Kingman	Arizona	All	5	4	33.0	13.0	519	2.8–7.3	11.7	20	1.5	25	1.050
Warangal	India	All	2	—	37.0	5.0	570	4	23.1	3	1.5	53	1.024
Richmond	Australia	Pilot	2	0	44.0	26.1	1,253	1.0–1.4	17.0	3	1.5	15	0.953
New Hanover	North Carolina	Pilot	4	10	135.0	64.0	750	3.3–4.2	19.4	2	1.5	5	1.082
Hamilton (meat)	New Zealand	Pilot	2	0	160.0	96.0	3,380	0.3–7.3	15.0	3	1.5	18	1.087
Median									**18.0**		**1.5**	**21.5**	**1.056**

error (transit time offset). Calibrations were performed for best estimates of the internal hydraulics for each wetland. Therefore, P-values range from 2 (New Hanover, measured $P = N = 2$) to near plug flow conditions, based upon system geometry. The value $C^* = 1.5$ was used when data did not allow determination of a calibrated value. However, calibrated values were with a narrow range, 0.8–2.2 mg/L. The median k_{20}-value for TN is 21.5 m/yr, but the range is wide.

Temperature coefficients had a median value of 1.056, indicating a relatively strong thermal effect on the suite of microbial processes that contribute to TN reduction.

The wetlands in Table 9.20 are all continuous flow systems. Four storm water wetlands in Sweden were used to model TN disappearance (Arheimer and Wittgren, 1994). Nitrate comprised about half of the total, which was in the range $5 \leq \mathrm{TN} \leq 20$ mg/L. The calibration spanned two calendar years. The model was a temperature sensitive, first-order areal model with a zero background concentration:

$$J_{\mathrm{TN}} = \left[r\, T_{10} \right] C_{\mathrm{TN}} \tag{9.47}$$

where

C_{TN} = total nitrogen concentration, mg/L = g/m^3
J_{TN} = total nitrogen removal flux, g/m^2·d
r = retention calibration factor, m/°C·d
T_{10} = mean temperature for the last ten days, °C

The product $[rT_{10}]$ is equal to the first-order irreversible rate constant (k) for TN reduction. The value of $r = 0.0023$ m/°C·d calibrated data from the four wetlands over two years with $R^2 = 0.92$. Over the range $5 \leq T \leq 25$°C, the equivalent values are $k_{20} = 16.2$ m/yr and $\theta = 1.081$.

Agronomic Wetlands (Lightly Loaded Systems)

When the TN loading to the wetland is less than the growth requirements of the plants and algae by a considerable margin, the removal of TN is very likely to be mediated by the growth and decay of biomass. As a rough guideline, this situation occurs for TN loadings less than approximately 120 gN/m^2·yr (Kadlec, 2005d). This occurs for over one-quarter (28%) of the 141 wetlands displayed in Figure 9.29. It is important to note that low inlet TN load very often means very low inlet TN concentration, close to background; and consequently, there is no ability to obtain meaningful calibrations of TN rate constants.

Uptake presumably occurs for both the ammonia and nitrate components of TN, and release may be considered to add to the organic component. As for TKN, plant uptake rates do not correspond to the annual cycle of water temperatures, and hence TN removal in agronomic wetlands cannot be characterized by modified Arrhenius θ-factors. The contrast between agronomic and microbial control of the TN rate constants is illustrated for New Hanover, North Carolina, and Listowel, Ontario, in Figure 9.34. The New Hanover TN loading was high, and the rate constants are seasonally synchronized with water temperature. But for Listowel, there is a high uptake in the spring growth period, which occurs at moderately cool temperatures, and no correspondence between monthly rate constants and temperature is present.

For the agronomic Listowel wetland, it would be appropriate to utilize monthly k-values:

$k_{\mathrm{TN},j}$ = 4.2	2.4	4.0	11.9	24.3	13.7 m/yr
j = 1	2	3	4	5	6
$k_{\mathrm{TN},j}$ = 7.1	4.4	7.0	11.6	14.0	6.9 m/yr
j = 7	8	9	10	11	12

(9.48)

In any case, it is prudent to examine the actual seasonal progression of k-values, to ascertain whether a θ-value approach is warranted or not. Additionally, if there is not a temperature correlation, the R^2 for a θ-factor regression will be very low.

Depth Effects

The parameters of first-order models are referred to as "rate constants," but there is no *a priori* reason to believe that these very empirical "constants" do not in fact depend upon

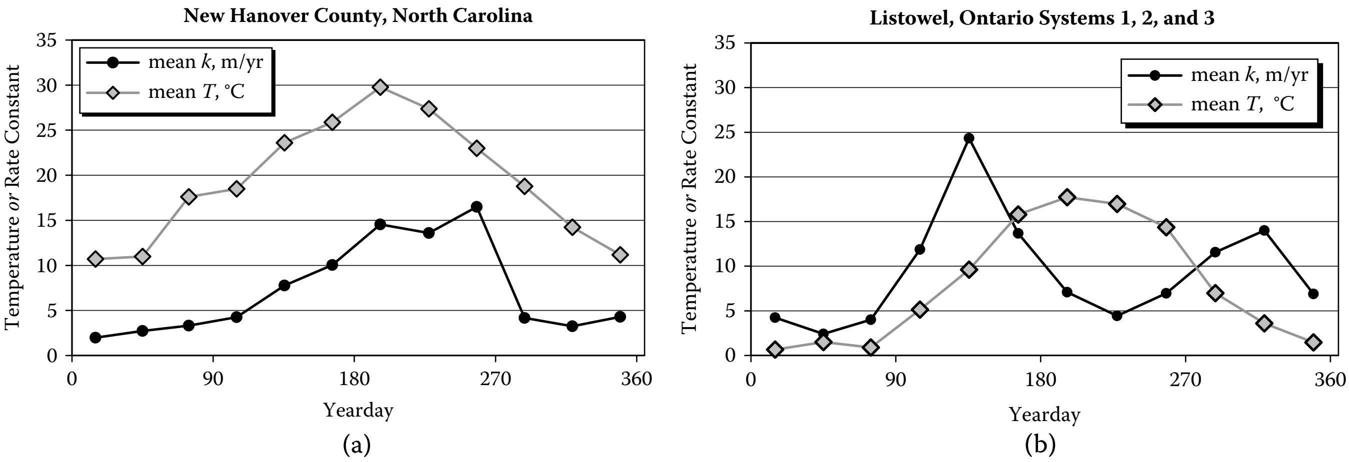


FIGURE 9.34 The relationships between temperature and total nitrogen rate constants for two FWS wetlands. The Listowel total nitrogen loading was 111 g/m^2·yr, and New Hanover was 750 g/m^2·yr. Data for four years are represented in each case.

other operational characteristics of the wetland. The design variable of depth is indirectly involved in sizing computations using areal rate constants, but is directly involved if volumetric rate constants are employed. If k-values change with depth, then that effect must be accounted in design. The relation $k = (\varepsilon h)k_V$ requires that both k and k_V cannot be independent of depth.

If k_V is constant with respect to depth, then k is proportional to depth. That condition implies the removal of TN to be uniformly distributed vertically throughout the water column. If k is constant, k_V is inversely proportional to depth. That condition corresponds to removal apportioned to wetland surface area. Neither ideal extreme is likely to be present in a treatment wetland, but data often show FWS wetlands to behave with constant k, meaning that k_V increases with decreasing depth. For instance, the reduction of TN in 17 side-by-side wetland cells at Jackson Bottoms, Oregon (SRI, 1990), shows k_V inversely proportional to depth (Figure 9.35). The values $P = \infty$ (very long slender wetlands) and $C^* = 1.5$ mg/L were presumed in calibration. The implications for design are very important. *If the volumetric model is utilized in FWS calculations, there appears to be the option of increasing performance by increasing the water depth, and hence increasing the nominal detention time. However, that advantage is lost if the volumetric rate "constant" decreases with increasing depth, as indicated in Figure 9.35.*

Total Nitrogen Rate Constants for HSSF Wetlands

Ranges in annual average k-rates for HSSF wetlands are summarized in Table 9.19, with the assumptions of PTIS = 6 and C^* = 1.0 mg/L. The median k-rate is 8.4 m/yr. However, there is wide variability in the data, and the 10th–90th percentile range is 1.9–30.5 m/yr.

There appears to be little, if any, temperature dependence on TN removal in HSSF wetlands. Period of record data from 16 HSSF wetlands are presented in Table 9.21. The median θ-factor was 1.005; and the 10th–90th percentile range is 0.990–1.029.

Intrasystem Variability

In some treatment systems, TN must be reduced to regulatory limits. Regardless of the design method for nitrogen removal, the method should ensure regulatory compliance. Models represent only the seasonal trends of effluent TN concentrations, leaving a considerable amount of probabilistic scatter in performance (see Chapter 6). Monthly limits are the most common averaging period for regulatory compliance. Therefore, it is useful to examine the variability of monthly average outlet concentrations, here represented by a fractional addition to the trend value (Ψ):

$$C_{trend} = C_{avg}\left[1 + A\cos\left(\omega(t - t_{max})\right)\right] + E \tag{6.1}$$

$$\Psi = \frac{E}{C_{trend}} \tag{6.60}$$

where

A = fractional amplitude of the seasonal cycle
C = instantaneous monthly outlet concentration, mg/L
C_{avg} = period of record average outlet concentration, mg/L
C_{trend} = cyclic mean concentration, mg/L
E = random portion of the outlet concentration, mg/L
t = time of the year, Julian day
t_{max} = time of the year for the maximum outlet concentration, Julian day
Ψ = fractional addition to the trend value, dimensionless

The multiplier on the trend value is $(1 + \Psi)$. The set of monthly averages for a wetland over some period of record will yield a distribution of Ψ-values. Because we are interested in preventing or controlling exceedances, the upper percentile points of that distribution are useful in design. For instance, the 90th percentile represents the fractional addition to the trend that may be expected to occur one time out of ten during the period of record. Note that these Ψ-values are not

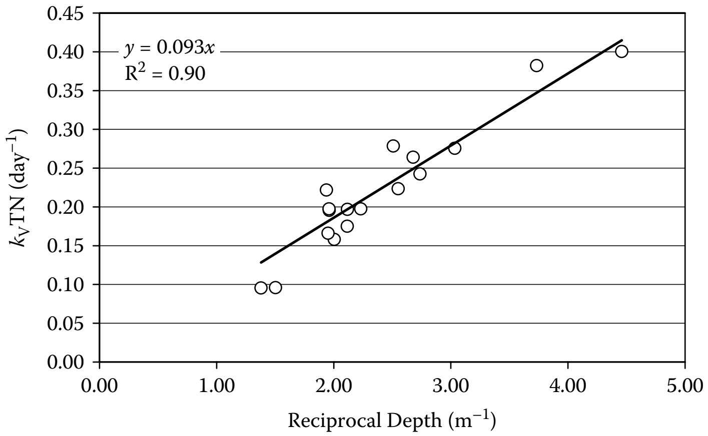


FIGURE 9.35 Depth effect on the Jackson Bottoms, Oregon, FWS total nitrogen volumetric rate "constant."

TABLE 9.21
Temperature Coefficients for Total Nitrogen Removal Rate Constants in HSSF Wetlands

Site	Reference	Cell	T range (°C)	Mean HLR (cm/d)	Mean C_i (mg/L)	Mean C_o (mg/L)	Theta
Richmond, NSW	Bavor *et al.* (1988)	Gravel	11–24	3.8	44.1	21.6	1.002
Richmond, NSW	Bavor *et al.* (1988)	*Typha*	11–24	4.6	44.1	21.7	1.001
Richmond, NSW	Bavor *et al.* (1988)	*Schoenoplectus*	11–24	5.1	44.1	20.1	1.007
Richmond, NSW	Bavor *et al.* (1988)	Mixed A	11–24	4.6	44.1	18.6	1.006
Richmond, NSW	Bavor *et al.* (1988)	Mixed B	11–24	3.8	44.1	15.3	0.996
Hardin, Kentucky	TVA, unpublished	1	6–27	9.7	20.2	13.2	0.956
Grand Lake, Minnesota	Unpublished	1	1–17	1.0	59.4	41.2	1.013
NERCC, Minnesota	Unpublished	1	1–16	1.4	87.6	57.1	1.017
NERCC, Minnesota	Unpublished	2	1–16	1.4	87.6	59.8	1.031
Lincoln, Nebraska	Vanier and Dahab (1997)	*Typha, Schoenoplectus*	4–21	9.5	34.8	23.0	1.026
North Yorkshire 1, U.K.	CWA database (2006)	1	4–15	4.5	36.4	31.3	0.995
Hamilton, New Zealand	Tanner *et al.* (1998b)	L1	10–25	1.5	55.9	18.6	1.062
Hamilton, New Zealand	Tanner *et al.* (1998b)	L2	10–25	2.5	55.9	20.7	1.003
Hamilton, New Zealand	Tanner *et al.* (1998b)	L3	10–25	3.3	55.9	24.1	1.000
Hamilton, New Zealand	Tanner *et al.* (1998b)	L4	10–25	4.9	55.9	28.1	1.007
Hamilton, New Zealand	Tanner *et al.* (1998b)	L5	10–25	6.9	55.9	31.0	0.985

Percentile	Theta
0.05	0.978
0.10	0.990
0.20	0.996
0.30	1.001
0.40	1.002
0.50	1.005
0.60	1.007
0.70	1.010
0.80	1.017
0.90	1.029
0.95	1.039

quite compliance percentiles, because of the finite number of months involved in a calibration set.

Table 9.22 lists percentiles of the monthly Ψ-distributions for some representative FWS wetlands. These data indicate that the median of the 95th percentile is an additional 70% above the trend, and that the median of the 90th percentile is an additional 55%.

To incorporate this variability into design, the wetland designer must oversize the wetland. Suppose it is necessary to meet a monthly limit of 3.0 mg/L of TN at least nine times out of ten (90th percentile). The designer should proceed in three steps:

1. Use seasonal variations, either temperature-driven for microbial wetlands or month-driven for agronomic wetlands, to establish the bottleneck period, which is likely to be winter unless there are large reductions of inflow rates and concentrations in winter. The month with the largest outlet TN is selected as the controlling month.
2. Apply a multiplier to account for the random part of the effluent concentration distribution during the controlling month. For instance, this

TABLE 9.22
Trend Multipliers for Effluent Total Nitrogen Concentrations in FWS Wetlands

System	Years of Data	Excursion Frequency 50%	80%	90%	95%
Brawley, California	4	0.95	1.48	1.79	2.03
Brighton, Ontario	4	1.05	1.18	1.21	1.24
Kingman, Arizona	5	0.97	1.60	1.79	2.56
Lakeland 1, Florida	7	0.83	1.24	1.49	1.60
Linköping, Sweden	3	0.98	1.21	1.28	1.35
Listowel 3, Ontario	4	0.86	1.29	1.65	1.72
Listowel 4, Ontario	4	0.94	1.25	1.62	1.89
New Hanover, North Carolina	4	0.97	1.20	1.43	1.71
Orlando Easterly Wetland, Florida	9	0.99	1.20	1.36	1.46
Tres Rios H1, Arizona	6	0.96	1.42	1.65	1.70
Median		**0.96**	**1.25**	**1.55**	**1.70**
Mean		**0.95**	**1.31**	**1.53**	**1.73**

Note: Data are approximately monthly. The multiplier on the trend concentration is (1 + Ψ); see Equation 6.61. For instance, one out of ten months, we can expect a total nitrogen concentration 1.55 times the long-term mean value based on the median of the 10 wetlands.

could be the median multiplier of 1.70 from Table 9.22.

3. Increase the wetland size so that the calculated seasonally variable effluent concentration does not exceed 3/(1.70) = 1.76 mg/L during the bottleneck period.

The reader should note that this correction is not a safety factor. It is known from the nature of existing data sets that excursions will occur. It becomes necessary to plan for those excursions, so that potential exceedances occur at less than the predetermined frequency. Wetland data may sometimes contain sufficient intrasystem variability to place the 100th percentile above the median inlet concentration. Thus, it may not be possible to design a wetland to totally avoid the possibility of monthly exceedances.

9.9 PERFORMANCE FOR AMMONIA

Ammonia is an intermediate in the sequential processing of nitrogen in treatment wetlands, which is produced by ammonification of organic nitrogen, and oxidized by aerobic and possibly anaerobic processes. Because of toxicity of un-ionized ammonia in receiving aquatic ecosystems, this nitrogen species is often singled out for regulation. The fraction of un-ionized ammonia depends upon water temperature as well as total dissolved ammonia, and hence regulation may be seasonal, with lower concentration limits in summer months.

Percent removal is an inadequate measure of ammonia performance. Indeed, negative removals may result, for two reasons. First, because ammonia is an intermediate in the processing sequence, production may exceed removal. This effect may be seen in the transect studies for the FWS wetland at Listowel, Ontario, in which ammonification produces ammonia in the inlet zones (Figure 9.36). Second, the analytical limits for the determination of ammonia in the laboratory may skew the difference between values at or below the selected detection limit, especially for low influent concentrations. As a result, 30 of 208 FWS wetlands reporting ammonia data show negative removals, and even higher fractions of systems in smaller databases have been reported to have negative removals (Reed *et al*., 1995).

REDUCTION OF AMMONIA IN FWS WETLANDS

Reductions of ammonia in treatment wetlands systems can be represented by loading analysis and the *P-k-C** model. The median net period-of-record removal rate for 118 FWS systems receiving more than 1 mg/L ammonia nitrogen is 127 g/m²·yr (Table 9.23). There is, however, wide variability among systems.

The load response graph for ammonia reflects the effluent ammonia concentration produced (C_o) by a given TKN LRI (= HLR × C_i) and ammonia concentration (C_i). Multiple data sets are represented by a trend that shows increasing C_o with increasing LRI, with different groupings associated with each inlet concentration (Figure 9.37). The overall slope of the intersystem data on the log–log coordinates is about 1.0 for all inlet concentrations. There is not an apparent lower limit to exit ammonia concentrations, indicating a near-zero background for this species. As for TKN and TN, inlet loading is an insufficient design specification, because hydraulic load and inlet concentration are not interchangeable factors in the load representation.

Implied Oxygen Supply in FWS Wetlands

Because all mechanisms of ammonia reduction require oxygen to varying degrees, it is useful to speculate on the amount consumed in the FWS wetland database. Most studies of ammonia removal in constructed wetlands assume the occurrence of the "classical" sequence of autotrophic nitrification followed by respiratory denitrification. The nitrification step requires 4.3 gO/gN, as well as an additional 0.3 gO/gN to supply the organisms.

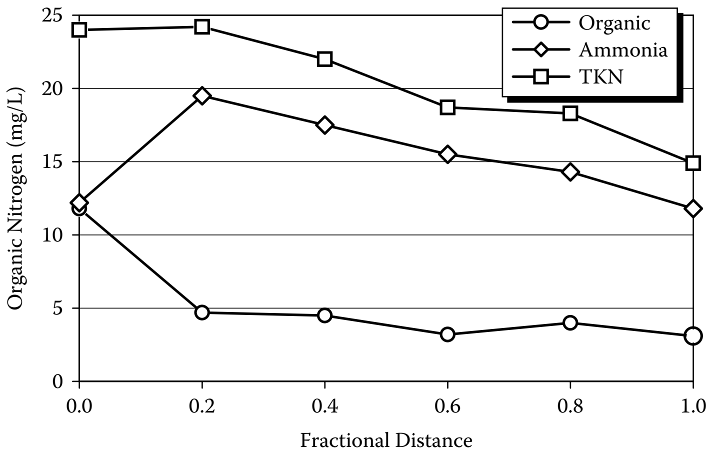


FIGURE 9.36 Profiles of major dissolved nitrogen species in Listowel, Ontario, FWS Wetland 4 in Summer 1984. Lines are model calibration, symbols denote data points which are averages of biweekly data over three months. (From Kadlec and Knight (1996) *Treatment Wetlands*. First Edition, CRC Press, Boca Raton, Florida.)

TABLE 9.23
Annual Reduction of Ammonia Nitrogen in FWS Wetlands

Stipulations

1. Data restricted to wetlands receiving inlet $C > 1$ mg/L ammonia nitrogen.
2. Period of record averages are used in calculations.
3. For k-value calculations, the following P-k-C^* parameters are selected:
   a. $C^* = 0$ mg/L
   b. $P = 3$ TIS
4. Sequential conversion of organic to ammonia nitrogen is accounted.
5. An annual vegetative nitrogen uptake of 40 g/m²·yr is apportioned to ammonia and nitrate according to their mean concentrations.
6. 90% of the vegetative uptake is recycled, and 10% permanently buried.

	HLR (cm/d)	Org-N In (mg/L)	Org-N Out (mg/L)	NH_4–N In (mg/L)	NH_4–N Out (mg/L)
Mean	7.3	10.5	4.9	75.3	45.5
Median	4.1	5.7	2.7	15.5	7.3
Max	110.0	69.5	29.6	405.5	279.3
Min	0.3	0.0	0.0	1.2	0.1

Results (N = 118 wetlands)

Percentile	Load Removed (g/m²·yr)	Rate Coefficient (m/yr)
0.0	7	0.4
0.1	37	4.7
0.2	51	7.0
0.3	68	8.7
0.4	85	11.1
0.5	127	14.7
0.6	233	26.1
0.7	387	45.1
0.8	823	59.2
0.9	1,941	85.6
1.0	4,680	258.5

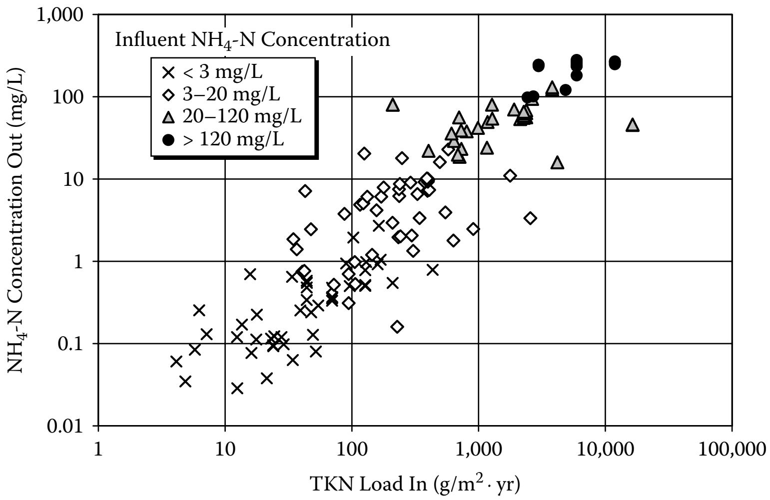


FIGURE 9.37 Load–concentration plot for ammonia nitrogen in FWS wetlands. Inlet TKN loading is used, because ammonification can add to the production of ammonia. Points are separated according to the inlet ammonia concentration range. Each point represents the entire period of record (POR) for one of 149 wetlands.

TABLE 9.24
Oxygen Usage in FWS Wetlands as a Function of Assumed Stoichiometry of BOD Reduction and Ammonia Removal

Percentile	BODLR[a] ($gO/m^2{\cdot}d$)	Internal ALR[b] ($gN/m^2{\cdot}d$)	External ALR[b] ($gN/m^2{\cdot}d$)	Maximum[c] O Usage ($gO/m^2{\cdot}d$)	Intermediate[c] O Usage ($gO/m^2{\cdot}d$)	Minimum[c] O Usage ($gO/m^2{\cdot}d$)
0.05	0.00	0.02	−0.31	−0.05	−0.02	−0.08
0.10	0.01	0.05	−0.19	0.09	0.04	−0.03
0.20	0.05	0.05	−0.10	0.35	0.17	0.01
0.30	0.16	0.07	−0.06	0.62	0.33	0.04
0.40	0.37	0.09	−0.03	0.98	0.52	0.10
0.50	0.86	0.12	0.02	1.47	0.91	0.19
0.60	1.12	0.16	0.20	3.65	1.89	0.53
0.70	1.93	0.25	0.48	6.92	3.92	1.02
0.80	2.80	0.34	1.14	16.72	7.59	3.39
0.90	5.17	0.67	4.88	43.14	17.22	8.79
0.95	24.11	0.86	6.90	50.30	27.44	12.10

Note: Both internal (from organic N) and external ammonia loads are considered. The Anammox route requires half the ammonia to be converted to nitrite, which needs approximately 1.7 gO/gN. BOD may be reduced by anaerobic or aerobic processes. The data represent period of record data from 95 wetlands.

[a] BODLR = BOD load removed.

[b] ALR = ammonia load removed.

[c] The maximum case assumes 1.5 gO/gBOD and 4.6 gO/gNH_4–N.
The intermediate case assumes 1.0 gO/gBOD and 1.7 gO/gNH_4–N.
The minimum case assumes 0.0 gO/gBOD and 1.7 gO/gNH_4–N.

As noted in Chapter 8, there is similar doubt that BOD reduction is solely due to microbial processes utilizing free oxygen, since anaerobic processes (such as fermentation) will also remove BOD. More accurate estimation would allow for settlement of solid BOD, and consumption in denitrification (Cooper, 1999). Nevertheless, if an upper limit to the necessary oxygen is sought, then CBOD reduction via aerobic degradation must be added to the requirement, and included in the overall oxygen demand.

Note: *The assumption of "classical" nitrification/denitrification coupled with assumption of aerobic degradation of BOD will always result in the highest estimate of the implied oxygen supply.*

The use of "alternate" nitrogen chemistry, coupled with different ranges of BOD degradation (aerobic or anaerobic) will result in estimates of the implied oxygen supply considerably lower than that associated with "classical" nitrification and denitrification. Table 9.24 shows the distribution of implied oxygen supply to FWS wetlands under several different stoichiometric assumptions.

As seen in Table 9.24, FWS wetlands are relatively oxygen-transfer limited systems, with a median implied transfer rate of 1.47 $gO/m^2{\cdot}d$ for the most optimistic stoichiometric assumption (1.5 gO/gBOD; 4.6 gO/gNH_4–N). However, there is a wide range in the data set; the 10th and 90th percentiles are 0.09–43.14 $gO/m^2{\cdot}d$ for the most optimistic stoichiometry. Values at the higher probability levels are presumably increased by factors such as wind mixing.

Reduction of Ammonia in HSSF Wetlands

The median annual-average removal rate for HSSF wetlands (213 system-years of data) is 208 $g/m^2{\cdot}yr$, as indicated in Table 9.25.

It is also useful to evaluate wetland performance (C_o) as a function of the inlet loading (see Figure 9.38). Figure 9.38 represents data from 112 HSSF wetlands (198 system-years). In general, there is an overall upward trend of the outlet ammonia concentration (C_o) in response to the inlet TKN loading, with a log–log slope of approximately 0.5. However, this apparent slope is due in a large measure due to the shift in inlet concentrations. When a particular inlet concentration group (like those shown in Figure 9.38) is considered, the change in outlet ammonia concentration is much less, as the intersystem slope for each concentration grouping is approximately 0.25. This has important design implications, because as the hydraulic loading to the wetland is decreased, the reduction in effluent concentration follows the slope of the inlet concentration group, not the overall data set. Use of the overall data set will overpredict the reductions in effluent ammonia concentrations as the hydraulic loading is decreased.

Plant Uptake of Ammonia

"Conventional wisdom" is that nitrification is the dominant mechanism for ammonia reduction in HSSF wetlands (U.S. EPA, 2000a; Crites *et al.*, 2006). However, plant uptake can have an effect in some cases. To place this in perspective, the growth of biomass during the year is assigned to be

TABLE 9.25
Annual Implied Nitrification of Ammonia Nitrogen in HSSF Wetlands

Stipulations

1. The decomposition of 2,000 g/m²·yr of biomass causes production of 36 gN/m²·yr of organic nitrogen.
2. Annual averages are used in calculations.
3. For *k*-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 0$ mg/L
   b. $P = 6$ TIS
4. Ranges of variables:

	HLR (cm/d)	NH_4–N In (mg/L)	NH_4–N Out (mg/L)
Mean	7.0	39.7	28.3
Median	4.7	24.0	16.1
Max	41.2	229.5	182.0
Min	0.1	0.3	0.1

Results (N·*t* = 213 wetland-years)

Percentile	Load Removed (g/m²·yr)	Rate Coefficient (m/yr)
0.05	−2	0.0
0.10	19	0.4
0.20	69	2.7
0.30	123	5.2
0.40	161	8.4
0.50	208	11.4
0.60	258	15.0
0.70	333	18.8
0.80	438	29.8
0.90	573	63.3
0.95	770	133.3

2,000 g/m²·yr (this has been shown to be a representative value in Chapter 3). At a tissue nitrogen of 2%, 40 gN/m²·yr are used, of which the large majority (approximately 90%) is eventually returned via decomposition. A large HSSF database has been examined for the proportions of plant uptake and nitrification. In this analysis, an allowance has been made for ammonification of organic nitrogen as an internal source of ammonia. It is found that in 25% of the 117 planted HSSF wetlands surveyed, plant uptake is at least a quarter of the annual nitrification rate. Thus, the designer needs to be aware that at low ammonia loading rates (the approximate critical level has earlier been suggested to be 120 gN/m²·yr), plant uptake may be a factor in the processing of ammonia.

Implied Oxygen Supply in HSSF Wetlands

Because all mechanisms of ammonia reduction require oxygen to varying degrees, it is useful to speculate on the amount consumed in the HSSF wetlands in the database. Most studies of ammonia removal in constructed wetlands assume the occurrence of the "classical" sequence of autotrophic nitrification followed by respiratory denitrification. The nitrification step requires 4.3 gO/gN, as well as an additional 0.3 gO/gN to supply the microorganisms.

The result is an "implied oxygen requirement" of the decrease in CBOD (aerobically) plus 4.6 times the nitrification rate. However, a number of authors have used the decrease in ammonia rather than the mass balance nitrification rate (Cooper, 1999; Platzer, 1999; Noorvee *et al.*, 2005b). This estimate is high because of the omission of plant uptake of ammonia, but low because of the internal ammonia production assignable to the loss of organic nitrogen. Here, the mass balance approach is used to estimate nitrification, which corrects for these effects. However, the artifact of alternative mechanisms of ammonia processing remains. The assumption of "classical" nitrification/denitrification, coupled with

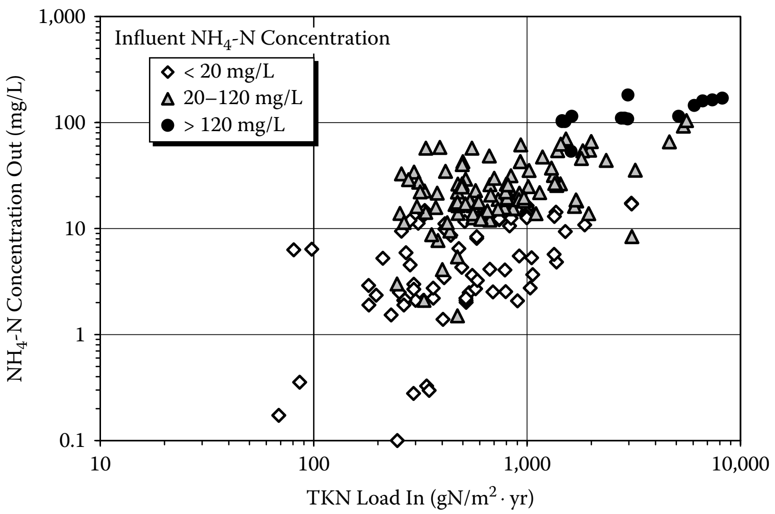


FIGURE 9.38 Load response data for HSSF wetlands. The loading includes organic nitrogen, because of the potential for ammonia production via ammonification. Annual average information from 112 wetlands and 198 wetland-years is shown.

assumption of aerobic degradation of BOD, will always result in the highest estimate of the implied oxygen supply.

However, nitrification–denitrification and organic carbon removal are closely coupled in treatment wetlands and nitrogen transformation intermediates (such as NO_2^- and NO_3^-) rarely accumulate in HSSF wetland beds. There is increasing evidence that in such oxygen-limited environments, nitrification, denitrification, and other microbial processes (e.g., methane oxidation) may be much more closely coupled (also described as integrated or simultaneous processes) and may include a range of alternative and co-metabolic pathways. Examples of potential alternative pathways with reduced overall oxygen requirements that have relevance to treatment wetlands have been discussed above, and include: oxygen-limited autotrophic nitrification–denitrification, and anaerobic ammonium oxidation (anammox). There is also the possibility of oxidation of ammonium by heterotrophs deriving energy from organic substrates.

The use of "alternate" nitrogen chemistry, coupled with different ranges of BOD degradation (aerobic or anaerobic), will result in estimates of the implied oxygen supply considerably lower than that associated with "classical" nitrification and denitrification. Table 9.26 shows the distribution of implied oxygen supply to HSSF wetlands under several different stoichiometric assumptions.

As seen in Table 9.26, HSSF wetlands are relatively oxygen-transfer limited systems, with a median implied transfer rate of 6.3 $gO/m^2{\cdot}d$ for the most optimistic stoichiometric assumption (1.5 gO/g BOD; 4.6 gO/gNH_4–N). However, there is a wide range in the data set; the 10th and 90th percentiles are 2.1–21.1 $gO/m^2{\cdot}d$ for the most optimistic stoichiometry, which is probably an overestimate in the case of HSSF wetlands. The use of "alternate" nitrogen stoichiometries results in much lower implied oxygen transfers.

Oxygen Transfer—Plants or Atmospheric Diffusion?

Nothing has been more controversial in the literature of wetland ammonia removal than the ongoing discussions of the role of plants in oxygen supply. One side of this issue is represented by the idea that physical oxygen transfer is negligible, and that plant oxygenation is responsible for the entirety of ammonia oxidation. The logical extension of this point of view is that the more roots, the more oxygen transfer. Accordingly, sources such as U.S. EPA (1993f), Reed *et al.* (1995), and Crites *et al.* (2006) have promulgated a universal root-volumetric oxygen transfer rate of 7.5 $gO/m^3{\cdot}d$, based upon data from the Santee, California, test facility. For a 45-cm rooting depth, and 100% coverage of the wetland, this amounts to an areal delivery of 3.4 $gO/m^2{\cdot}d$. This concept places a premium on both deep, extensive rooting and healthy, complete coverage. The Santee study (Gersberg *et al.*, 1986) found that *Phragmites* and bulrushes "remained very healthy throughout our study," but cattails "showed a marked yellowing, and most had died after six months."

McGechan *et al.* (2005a, b) modeled six horizontal layers in a HSSF system in West Harwood, Scotland, and allowed for both atmospheric aeration and plant aeration. Although it was possible to reproduce the longitudinal profiles of nitrogen species, they could not accurately partition the oxygen supply between the two routes. The implied oxygen supply was 8 $gO/m^2{\cdot}d$.

Table 9.27 shows the results of side-by-side studies with planted and unplanted HSSF wetlands. On average, the

TABLE 9.26
Oxygen Usage in HSSF Wetlands as a Function of Assumed Stoichiometry of BOD Reduction and Ammonia Removal

Percentile	BODLR[a] ($gO/m^2{\cdot}d$)	Internal ALR[b] ($gN/m^2{\cdot}d$)	External ALR[b] ($gN/m^2{\cdot}d$)	Maximum[c] O Usage ($gO/m^2{\cdot}d$)	Intermediate[c] O Usage ($gO/m^2{\cdot}d$)	Minimum[c] O Usage ($gO/m^2{\cdot}d$)
0.05	0.36	–0.01	–0.14	0.9	0.6	0.0
0.10	0.72	0.05	–0.04	2.1	1.2	0.2
0.20	1.11	0.11	0.01	3.6	1.7	0.4
0.30	1.40	0.15	0.14	4.2	2.2	0.6
0.40	1.88	0.20	0.24	5.2	2.7	0.8
0.50	2.17	0.23	0.28	6.3	3.2	1.0
0.60	2.93	0.29	0.38	8.5	4.4	1.2
0.70	4.22	0.35	0.57	10.6	5.4	1.5
0.80	5.28	0.47	0.73	12.8	7.5	2.0
0.90	10.58	0.63	1.09	21.1	12.8	2.6
0.95	18.54	0.78	1.52	38.2	23.0	3.6

Note: Both internal (from organic N) and external ammonia loads are considered. The Anammox route requires half the ammonia to be converted to nitrite, which needs approximately1.7 gO/gN. BOD may be reduced by anaerobic or aerobic processes. The data represent 85 wetlands and 168 wetland-years of data.

[a] BODLR = BOD load removed.
[b] ALR = ammonia load removed.
[c] The maximum case assumes 1.5 gO/gBOD and 4.6 gO/gNH_4–N.
The intermediate case assumes 1.0 gO/gBOD and 1.7 gO/gNH_4–N.
The minimum case assumes 0.0 gO/gBOD and 1.7 gO/gNH_4–N.

TABLE 9.27
Maximum Implied Oxygen Requirements for Side-by-Side Studies of HSSF Wetlands with and without Plants

Reference	Site Name	System Name	Cell	Year	BODLR (g/m²·d)	Nitrification (g/m²·d)	Total Oxygen Implied	Unplanted/Planted Ratio
Theis and Young (2000)	Minoa, New York	E2.1A	U	1996–97	4.4	2.2	6.6	—
Theis and Young (2000)	Minoa, New York	E1.2A	S	1996–97	4.4	1.9	6.3	1.045
Theis and Young (2000)	Minoa, New York	E1.1A	P	1996–97	4.4	2.5	6.9	0.956
Theis and Young (2000)	Minoa, New York	E2.2A	P	1996–97	4.4	1.9	6.3	1.058
Akratos and Tsihrintzis (2007)	Thrace, Greece	MG-Z	Control	2004–06	3.3	0.99	4.3	—
Akratos and Tsihrintzis (2007)	Thrace, Greece	MG-C	Cattail	2004–06	3.4	2.03	5.4	0.789
Akratos and Tsihrintzis (2007)	Thrace, Greece	MG-R	Reed	2004–06	3.2	1.65	4.9	0.872
Bavor *et al.* (1988)	Richmond, NSW	—	Gravel	1984–86	2.4	5.4	7.7	—
Bavor *et al.* (1988)	Richmond, NSW	—	*Typha*	1984–86	2.1	4.9	6.9	1.116
Bavor *et al.* (1988)	Richmond, NSW	—	*Schoenoplectus*	1984–86	2.4	5.6	8.0	0.974
Bavor *et al.* (1988)	Richmond, NSW	—	Mix A	1984–86	2.2	5.7	7.9	0.981
Bavor *et al.* (1988)	Richmond, NSW	—	Mix B	1984–86	1.8	5.2	7.0	1.104
Tanner (1996)	Hamilton, New Zealand	1	Unplanted	1990–91	1.6	1.1	2.7	—
Tanner (1996)	Hamilton, New Zealand	1	Bulrushes	1990–91	2.2	2.5	4.7	0.581
Tanner (1996)	Hamilton, New Zealand	2	Unplanted	1990–91	1.6	1.2	2.8	—
Tanner (1996)	Hamilton, New Zealand	2	Bulrushes	1990–91	1.5	1.6	3.0	0.926
Tanner (1996)	Hamilton, New Zealand	3	Unplanted	1990–91	1.0	0.6	1.6	—
Tanner (1996)	Hamilton, New Zealand	3	Bulrushes	1990–91	1.1	1.2	2.3	0.703
Tanner (1996)	Hamilton, New Zealand	4	Unplanted	1990–91	0.9	0.6	1.5	—
Tanner (1996)	Hamilton, New Zealand	4	Bulrushes	1990–91	1.0	1.0	2.0	0.787
Van Oostrom and Cooper (1990)	Hamilton Horotiu	—	Gravel	1988–89	8.8	5.8	14.7	—
Van Oostrom and Cooper (1990)	Hamilton Horotiu	—	*Schoenoplectus*	1988–89	10.2	6.8	17.0	0.862
Van Oostrom and Cooper (1990)	Hamilton Horotiu	—	*Glyceria*	1988–89	9.4	5.8	15.2	0.963
Van Oostrom and Cooper (1990)	Hamilton Horotiu	—	Gravel	1989–90	5.1	3.5	8.6	—
Van Oostrom and Cooper (1990)	Hamilton Horotiu	—	*Schopenoplectus*	1989–90	5.3	3.8	9.2	0.938
Van Oostrom and Cooper (1990)	Hamilton Horotiu	—	*Glyceria*	1989–90	5.0	3.7	8.7	0.991
Gersberg *et al.* (1984)	Santee, California	G	—	—	3.8	1.0	4.8	—
Gersberg *et al.* (1984)	Santee, California	CT	—	—	4.5	1.9	6.4	0.750
Gersberg *et al.* (1984)	Santee, California	BR	—	—	5.3	5.4	10.7	0.449
Gersberg *et al.* (1984)	Santee, California	Phrag	—	—	4.5	4.5	9.0	0.532
Mean					**3.69**	**3.09**	**6.78**	**0.869**
Standard deviation					**2.47**	**1.99**	**3.96**	**0.188**

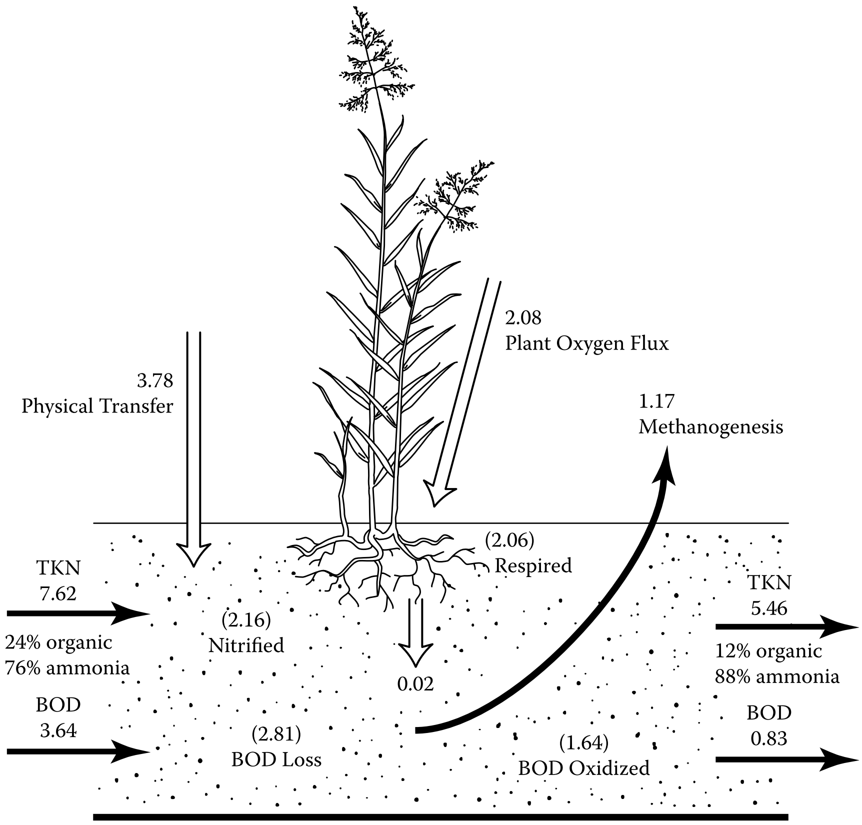


FIGURE 9.39 Fluxes of oxygen sources and sinks in a HSSF wetland, the latter expressed as oxygen equivalents. See text for explanation. Reinterpretation of Brix (1990) data from Kalø, Denmark.

unvegetated systems have implied oxygen transfers that are 87% of those observed for vegetated systems. The Gersberg *et al*. (1986) data stand out as the most influenced by plants, with 45–75% of the transfer ascribable to plants, depending upon which plant is considered. These studies show a mean implied oxygen transfer of 6.8 $gO/m^2 \cdot d$, apportioned equally to BOD and ammonia reduction.

A widely quoted study of Brix (1990) produced field measurements of oxygen transfer in the reed bed at Kalø, Denmark, and sought to explain the fate of that oxygen via independent measurements. Figure 9.39 shows a reinterpretation of that work. Field and laboratory measurements showed that about twice as much oxygen was supplied physically as via plants, and that of the plant oxygen flux, almost none was excess reaching the water column. Brix (1990) considered that BOD was the only sink, but here it is presumed that nitrification of ammonia is also a factor. The oxygen equivalent of BOD is assumed to be 1.0 gO/gBOD, and 4.6 gO/gBOD for TKN.

Field measurements of methane emission suggested that nearly half of the BOD loss was via anoxic routes (Brix, 1990) rather than oxidation, and here methanogenesis is used to close the oxygen supply–demand balance. These studies were conducted in spring, and it is presumed that any nitrogen required for early plant growth came from translocation. This study demonstrated that net plant oxygen transfer (from the roots to the water column) is a minor contributor in HSSF wetlands, and that the implied oxygen requirement (based on classical nitrification and aerobic degradation of BOD) is likely to be an overestimate because of observed anaerobic BOD reduction processes. *The reinterpretation here suggests that ammonification of organic nitrogen, and the implied oxygen requirement for that extra ammonia, are of equal or greater importance than the apparent change in influent–effluent ammonia concentrations.*

Wu *et al*. (2001) measured oxygen fluxes in sealed HSSF wetland mesocosms, both vegetated and unvegetated. Results were consistent with the "layer" concept of Figure 9.40, meaning that ammonia diffusion to the top aerobic zone was suggested to be the controlling mechanism. Importantly, there was but slight increase in oxygen utilization by vegetated systems, of 0.45 $gO/m^2 \cdot d$, for both 10 and 50 mg/L of ammonia nitrogen. The utilization was 6.0 and 7.5 $gO/m^2 \cdot d$, respectively. Larger, field mesocosm results confirmed the ammonia removal rate to be strongly increasing with ammonia concentration.

Reduction of Ammonia in VF Wetlands

Based on the information currently available, vertical flow wetlands are effective in oxidizing organic and ammonia-nitrogen. However, these removals are highly dependent on the mass loading of the wetland, and the operational regime (loading and resting periods) employed in the operation and

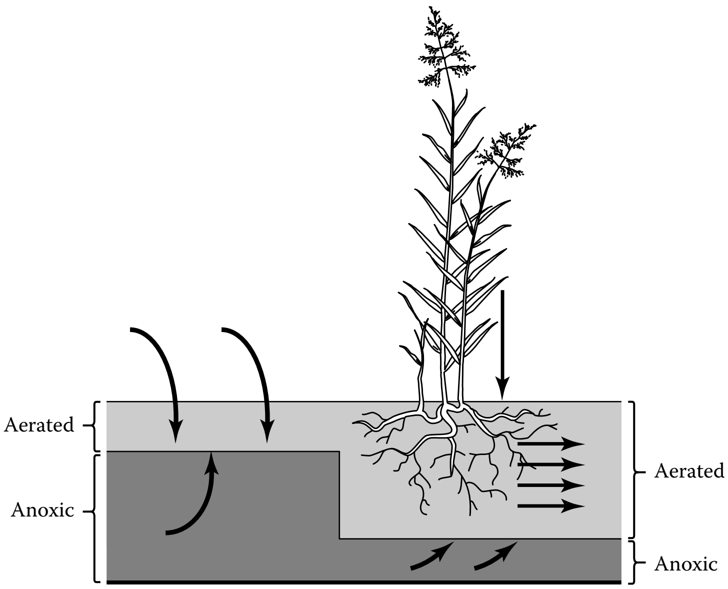


FIGURE 9.40 Simplified model of movement of oxygen into the HSSF bed.

maintenance of the system. It is also apparent that the cation exchange capacity (CEC) of the bed materials can play a large role in retention of ammonia and subsequent processing within the VF wetland bed (Johns *et al.*, 1998; Gisvold *et al.*, 2000; Austin *et al.*, 2006b).

Implied Oxygen Supply in VF Wetlands

The TN entering vertical flow wetlands can be compared to the TN leaving the system. Presumably, the reduction can be contributed to the oxidation and reduction of nitrogen within the wetland. The different chemical stoichiometries currently known provide a bound on the maximum and minimum oxygen transfers that occur in pulse-loaded vertical flow beds (Table 9.28). It should be noted that pulse-loaded vertical flow wetlands appear to provide oxygen transfers considerably higher than FWS or HSSF wetlands (Tables 9.24 and 9.26).

Oxygen transfer is a key issue in the design of vertical flow wetland beds (Johansen and Brix, 1996; Cooper, 1999; Cooper *et al.*, 1999), and is discussed in more detail in Chapters 20 and 21.

Background Concentrations of Ammonia

There is not an apparent lower limit to exit ammonia concentrations in Figures 9.37 and 9.38, indicating a near-zero background for this species. This result has also been observed in natural treatment wetlands. For instance, transect water chemistry has been acquired at the Houghton Lake treatment wetland over a 28-year period, including measurements of ammonium nitrogen. These data show an exponential decrease in NH_4–N concentrations with distance from the discharge, thus supporting a first-order model. Summer season operation shows very low background concentrations (approximately 0.05 mg/L), both for the wetland prior to wastewater discharges and for the unaffected zones of the present-day wetland. Wastewater concentrations are much higher (approximately 10 mg/L), therefore it is accurate to represent the data with a first-order model with a zero background. Time series data for batch wetlands also support the near-zero background (Figure 9.41). The detection limit for ammonia is typically 0.05 mg/L, and thus the zero background usually represents some value less than the detection limit. The assumption of $C^* = 0$ mg/L appears to approximate operating data from all types of treatment wetlands.

Rates and Rate Constants

In conventional activated sludge treatment system design, removal of ammonia is often modeled with a Monod formulation, with a half-saturation constant of 1.0 mg/L (U.S. EPA, 1993b):

$$J_{AN} = \frac{k' C_{AN}}{K + C_{AN}} \tag{9.49}$$

where

C_{AN} = ammonia nitrogen concentration, mg/L (= g/m^3)
J_{AN} = ammonia nitrogen removal flux, g/m^2·yr
k' = Monod rate constant, m/yr
K = half-saturation constant, mg/L

This formulation has been adopted in some analyses of ammonia in wetlands (Langergraber, 2001), but the half-saturation constant is so low that near-zero-order behavior should be seen for $C_{AN} >> 1$. Instead, for FWS wetlands, there is a linear correlation between the annual removal flux and the annual mean ammonia concentration for 98 wetlands

TABLE 9.28
Oxygen Usage in VF Wetlands as a Function of Assumed Stoichiometry of BOD Reduction and Ammonia Removal

Percentile	BODLR[a] ($gO/m^2 \cdot d$)	Internal ALR[b] ($gN/m^2 \cdot d$)	External ALR[b] ($gN/m^2 \cdot d$)	Maximum[c] O Usage ($gO/m^2 \cdot d$)	Intermediate[c] O Usage ($gO/m^2 \cdot d$)	Minimum[c] O Usage ($gO/m^2 \cdot d$)
0.05	2.4	–0.2	0.1	14.2	7.6	1.0
0.10	2.6	–0.1	0.2	19.5	10.4	1.4
0.20	3.6	0.2	0.6	20.7	11.5	2.0
0.30	6.5	0.3	0.8	22.6	12.4	2.3
0.40	8.4	0.4	1.0	22.9	12.6	2.6
0.50	9.4	0.5	1.3	24.7	13.4	3.5
0.60	10.4	0.6	1.9	29.3	14.7	4.1
0.70	11.3	0.6	2.0	30.7	16.5	5.1
0.80	12.7	0.8	2.7	39.9	20.0	9.1
0.90	17.7	1.3	8.4	51.2	21.9	14.1
0.95	19.5	1.8	10.1	55.6	27.4	16.8

Note: Both internal (from organic N) and external ammonia loads are considered. The Anammox route requires half the ammonia to be converted to nitrite, which needs 3.43 gO/gN. BOD may be reduced by anaerobic or aerobic processes. The data represent 22 wetlands and 34 wetland-years of data.

[a] BODLR = BOD load removed.

[b] ALR = ammonia load removed.

[c] The maximum case assumes 1.5 gO/gBOD and 4.6 gO/gNH_4–N.
The intermediate case assumes 1.0 gO/gBOD and 1.7 gO/gNH_4–N.
The minimum case assumes 0.0 gO/gBOD and 1.7 gO/gNH_4–N.

for $C_{AN} > 1$, with $R^2 = 0.61$ and a k-value of $k_{AN} = 9.2$ m/yr. This result should not be surprising, because it has been noted that both ammonia volatilization and plant uptake may be involved in wetland ammonia removal, as well as microbial processes. Therefore, the area-based first-order expression is preferred to model the disappearance of ammonia nitrogen (nitrification and other processes combined) in wetland treatment systems:

$$J_{AN} = k_{AN} C_{AN} \tag{9.50}$$

where
k_{AN} = removal rate constant for ammonia N, m/yr

Ammonia Rate Constants for FWS Wetlands

Results across systems for the value $P = 3$ are given here. The value $C^* = 0.0$ mg/L is used, and the remaining model parameter is the k-value, selected to fit the model:

$$\frac{C_{AN,\,out}}{C_{AN,\,in}} = \left(1 + \frac{k_{AN}}{3q}\right)^{-3} \tag{9.51}$$

Parameter estimation is not reliable for low inlet concentrations, and those wetlands with $C_{AN,in} < 1$ mg/L have been excluded from calibration. Out of 147 wetlands with data for all species of nitrogen (see Figure 9.37), 118 met this criterion.

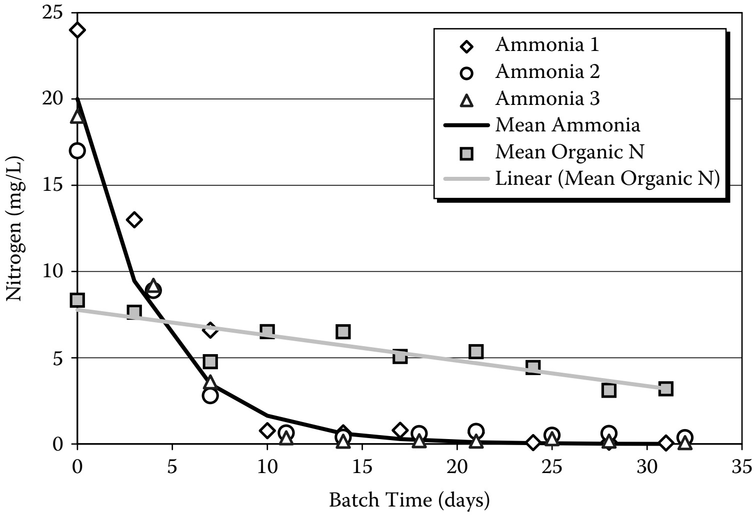


FIGURE 9.41 Reduction of ammonia nitrogen in the batch treatment FWS wetland systems at Humboldt, Saskatchewan. (Data from Lakhsman (1981) *A demonstration project at Humboldt to provide tertiary treatment to the municipal effluent using aquatic plants*. SRC Publication No. E-820-4-E-81. 74 pp. Saskatchewan Research Council.)

Calibration included ammonification (production) and nitrification (destruction), as well as return of organic nitrogen from the decomposition of biomass. The median annual rate constant was k_{AN} = 14.7 m/yr (Table 9.23). The 10th–90th percentile range is 4.7–85.6 m/yr. There is a significant temperature dependence of ammonia *k*-values. Even on an average annual basis, temperature or season may be an important determinant of the rate constant, and these factors are thus responsible for some of the intersystem variability in annual *k*-values. Accordingly, it is necessary to examine intra-annual effects.

Microbially Dominated Wetlands

Ammonia nitrogen is a nutrient source to support growth in the wetland. As for TKN and TN, when the ammonia loading to the wetland exceeds the growth requirements of the plants, bacteria, and algae by a considerable margin, the removal of ammonia is very likely to be microbially mediated. The loading limit for bacteria to predominate is approximately 120 gN/m^2·yr (Kadlec, 2005d). That means that slightly under half (47%) of the 118 FWS wetlands qualified as microbially controlled. In this ammonia data set, some wetlands are presumed to derive growth nitrogen from nitrate as well as ammonia.

In those cases where there is a significant contribution of ammonification to the ammonia loading, a sequential correction has been made to the kinetic scheme. However, there are many situations in which there is little ammonification, because incoming and outgoing organic nitrogen are low compared to ammonia. In the latter cases, the concentrations of ammonia alone suffice to estimate rate constants.

Results of calibration of k_{AN}-values for entire periods of record for representative wetlands are summarized in Table 9.29 Monthly averages were used to avoid synoptic error (transit time offset). Calibrations were performed for best estimates of the internal hydraulics for each wetland. Therefore, *P*-values range from 2 (New Hanover, measured $P = N = 2$) to near plug flow conditions, based upon system geometry. The median k_{20}-value for ammonia is 14.2 m/yr, but the range is wide.

There are strong seasonal effects on the ammonia rate constant, which for microbially mediated systems is in synchrony with the water temperature. A temperature coefficient (θ), is capable of accounting for these effects (see Chapter 6). Temperature coefficients had a median value of 1.049, indicating a relatively strong thermal effect on the suite of microbial processes that contribute to ammonia reduction.

Some of the effects of other environmental factors may be sought from the data in Table 9.29. The maximum implied oxygen requirement for complete nitrification has been calculated as 4.6 times the ammonia load removal (theoretical oxygen). This theoretical amount should be supplied by incoming DO together with reaeration. The Musselwhite system stands out as an anomaly, because the ammonia disappearance by traditional nitrification (19.7 gO/m^2·d) requires too much oxygen (see Chapter 5). Accordingly, Bishay and Kadlec (2005) have speculated on other mechanisms of ammonia removal in that wetland. Wetlands receiving very high ammonia also would require considerable theoretical oxygen (4–6 gO/m^2·d), which is at the upper, doubtful end of the supply range. Those wetlands may be losing small amounts of ammonia to volatilization (see Poach *et al*., 2002, 2004, and the discussion earlier in this chapter).

Open Water Zones

U.S. EPA (2000a) speculates that oxygenation is better provided by zones of open water, containing submerged vegetation if possible. Such zones are then incorporated as a necessary part of the wetland system design. We here observe that such reasoning would apply only to those FWS wetlands (about half) that are under microbial control of ammonia removal. In Table 9.29, there are five wetlands that had large fractions of open water (Pontotoc (2), Brawley, Oxelösund, Hassleholm). These display a wide range of k_{AN20} values, from quite low (6.9 m/yr) to quite high (140 m/yr). Consequently, there is nothing in this detailed data analysis to support the open water advantage concept. Kadlec (2005e) has drawn a similar conclusion for a wider set of FWS treatment wetlands. Open water zones are likely to foster a greater component of algal uptake for agronomic (lightly loaded) wetlands, in which ammonia removal is controlled by vegetation and algae.

Both algae and macrophytes are effective nitrogen cyclers, with the former dominating lagoon treatment systems, and the latter dominating FWS wetlands. Ammonia areal loadings to ponds are typically in the range defined here for microbial control. Algal biomass densities are typically lower than those for emergent macrophytes, but turnover times are much faster. A source of information on ammonia loss rates may be found in the literature on facultative ponds. For instance, Pano and Middlebrooks (1982) analyzed data from several ponds, and calibrated a first-order areal, well-mixed model to some of the data; and verified the model with the remainder of the data. The precise mechanism for ammonium reduction was not studied, but the model was chosen based on previous modeling of ammonia stripping ponds. Predictions were for k_{AN20} are 5–13 m/yr, for pH in the range of 7.0–7.5, which is the typical range for FWS treatment wetlands. The temperature coefficients derived from the Pano and Middlebrooks (1982) model are in the range θ = 1.045–1.069, for pH in the range of 7.0–7.5.

The Pano and Middlebrooks (1982) model remains as the benchmark reference at the present time (Abis, 2002; Mara, 2003). The same model was adopted by Soares *et al*. (1996), who found k_{AN} = 14 m/yr at 21°C, and 20 m/yr at 22°C, both for pH = 7.5. Zimmo *et al*. (2004) measured the individual processes in ponds, and found sedimentation (most), microbial (some), and volatilization (least) losses all played a part in removal. Azov and Tregubova (1995) found evidence for major amounts of sequential nitritation and nitrification, but little denitrification. Therefore, pond data probably represent mixed control mechanisms.

Because these pond areal k_{AN} values correspond to the central tendency of the FWS wetland values, there appears

TABLE 9.29
Dependence of Ammonia Rate Constants on Temperature for FWS Systems

Name	Location	Cells	Influent Type	Years	HLR In (cm/d)	NH_4–N In (mg/L)	NH_4–N Out (mg/L)	ALI[a] (g/m²·yr)	*P* (TIS)	Temp (°C)	k_{AN20} (m/yr)	Theta	Theoretical Oxygen (g/m²·d)	Vegetation
Brawley	California	All	River	4	8.9	4.1	0.1	131	∞	21.0	140.0	1.072	1.59	Open
Hassleholm	Sweden	All	Secondary	5	4.8	7.2	4.6	126	3	6.0	43.3	1.069	0.56	Open
Columbia	Missouri	All	Secondary	3	13.4	7.8	5.9	373	3	14.5	10.0	1.044	1.14	—
Brighton	Ontario	All	Lagoon	4	5.0	11.0	8.8	190	3	9.4	12.5	1.053	0.48	—
Musselwhite	Ontario	All	Mine	4	51.8	12.9	3.5	2,144	∞	11.0	245.0	1.033	19.70	—
Oxelösund	Sweden	All	Secondary	5	2.1	18.0	11.9	140	3	6.0	8.1	1.056	0.59	Open
Sacramento	California	Cell 7	Secondary	2	6.2	18.2	10.3	411	∞	15.9	9.2	1.109	2.25	—
Warangal	India	All	Primary	2	4.3	19.9	3.6	311	3	23.1	66.0	1.027	3.20	—
Linköping	Sweden	All	Secondary	3	4.2	26.5	9.1	255	3	8.6	17.8	1.032	2.12	—
Saginaw	Michigan	All	Leachate	2	1.6	29.5	5.0	183	3	19.3	28.5	1.045	1.92	—
Pontotoc	Mississippi	2	Animal	2	1.5	112.2	38.9	630	3	20.11	6.9	1.018	5.18	Open
Pontotoc	Mississippi	1	Animal	2	1.3	112.7	35.8	540	3	18.12	6.8	1.005	4.65	Open
New Hanover	North Carolina	All	Leachate	4	1.5	117.4	56.9	639	2	19.4	4.9	1.090	4.15	—
Duplin County	North Carolina	All	Animal	7	1.1	120.2	20.8	538	3	14.0	15.9	1.063	5.60	—
Median								**342**			**14.2**	**1.049**		

[a] ALI = influent ammonia loading rate

TABLE 9.30
Annual Rate Constants for Microbial FWS Wetlands with Appreciable Components of Submerged Aquatic Vegetation

Name	Location	Vegetation	Nonemergent Coverage (%)	T (°C)	TN In (mg/L)	TN Out (mg/L)	ALI[a] ($g/m^2 \cdot yr$)	*P*TIS	k_{AN} (m/yr)
Snogeröd	Sweden	*Elodea canadensis*	100	8	9.8	7.5	260[b]	∞	52
Durham	New Hampshire	*Elodea nuttallii*	100	17	20	11	220[b]	1	110
Texel	The Netherlands	*Elodea nuttallii* *Ceratophyllum demersum*	50	10	1.73	1.35	490	12	71
Texel	The Netherlands	*Elodea nuttallii* *Ceratophyllum demersum*	50	10	1.73	1.54	450	12	42
Arcata	California	*Potamogeton pectinatus*	73	13	7.4	6.3	313	3	7
Arcata	California	*Potamogeton pectinatus*	83	13	6.3	5	349	3	13
Arcata	California	*Potamogeton pectinatus*	78	13	5	4.1	162	3	6

[a] ALI = influent ammonia loading rate.
[b] Total nitrogen loading.

to be neither an advantage nor a disadvantage to unvegetated open water zones in wetlands designed for microbial ammonia reduction. However, submerged aquatic vegetation (SAV) zones may provide ammonia removal benefit, presumably due to subsurface oxygenation by these submerged photosynthesizers. On an annual basis, system data provided higher than the global median microbial ammonia rate constants in New Hampshire (Bishop and Eighmy, 1989), Sweden (Gumbricht, 1993a), and the Netherlands (Toet, 2003); but less than the than the global median microbial ammonia rate constants at Arcata, California (U.S. EPA, 1999) (Table 9.30). Effective strategies for maintaining SAV communities remain elusive, because they may easily be shaded out by floating plants (*Lemna*), overrun by aggressive semifloating species (*Hydrilla*), damaged by dry-out or extreme wind (hurricanes), or killed by high turbidity.

Agronomic Wetlands (Lightly Loaded)

When the ammonia loading to the wetland is less than the growth requirements of the plants and algae by a considerable margin, the removal of ammonia is very likely to be mediated by the growth and decay of biomass, including plants and algae. As stated above, a rough guideline is TKN loading less than approximately 120 $gN/m^2 \cdot yr$ (Kadlec, 2005d). If it is presumed that ammonia is the preferred form of nutrient nitrogen, this criterion may also be applied for ammonia.

Plant uptake rates do not correspond to the annual cycle of water temperatures, and hence ammonia removal in agronomic wetlands cannot be characterized by modified Arrhenius θ-factors. The contrast between water temperature sequences and ammonia rate constants is illustrated for Estevan, Saskatchewan, in Figure 9.42. The ammonia loading to this wetland was 28 $g/m^2 \cdot yr$, average during the growing season. No correspondence between monthly rate constants and temperature is present. For this agronomic wetland, it would be appropriate to utilize monthly k-values:

$$\begin{array}{llllllll} k_{TN,j} & =17.1 & 12.5 & 11.7 & 9.0 & 12.0 & 14.6 & 9.1\,\text{m/yr} \\ j & =5 & 6 & 7 & 8 & 9 & 10 & 11 \end{array}$$

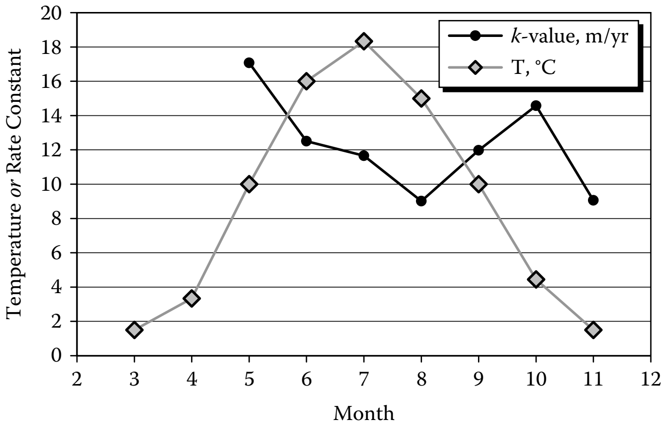


FIGURE 9.42 Seasonal ammonia rate constants for the Estevan, Saskatchewan, FWS treatment wetland. This system was operated seasonally (May–November) for nine years.

TABLE 9.31
Partitioning of Annual Ammonia Removal Rate Constants in FWS Wetlands According to the Control Mechanism

	Agronomic Wetlands (N = 73)		Microbial Wetlands (N = 74)	
Percentile	Inlet Load ($g/m^2 \cdot yr$)	Rate Coefficient, k_{AN} (m/yr)	Inlet Load ($g/m^2 \cdot yr$)	Rate Coefficient, k_{AN} (m/yr)
0.0	0.3	0.4	123	3.5
0.1	1.9	10	170	4.6
0.2	3.6	29	259	5.7
0.3	7.3	45	367	7.4
0.4	15.4	52	465	9.1
0.5	25.2	67	920	11.0
0.6	38.8	87	1,648	13.4
0.7	40.2	120	1,984	16.1
0.8	70.0	197	2,896	27.8
0.9	81.6	319	5,791	74.2
1.0	108.0	1,230	11,583	429

Similar patterns occur for other agronomic systems, such as Listowel, Ontario, 1, 2, and 3; and Arcata, California, enhancement wetlands.

It is noteworthy that first-order areal ammonia removal rate constants are higher for agronomic systems than for microbial systems. The median annual k_{AN} for agronomic wetlands is 67 m/yr ($N = 73$), while for microbial wetlands it is 11 m/yr ($N = 74$). Stated another way, 80% of agronomic wetlands have annual $k_{AN} > 30$ m/yr, whereas 80% of microbial wetlands have annual $k_{AN} < 30$ m/yr (Table 9.31). Therefore, the designer must be cognizant of the potential for a change in the type of ammonia removal as prospective loadings are varied, and the system shifts from plant-dominated (agronomic) to microbially dominated.

It is also important to recognize that agronomic control does not mean that the ultimate removal of ammonia is to the biomass compartment of the wetland. It has been estimated that 90% of uptake is recycled to the water column, presumably during periods of warmer temperatures. It may be speculated that the return of nitrogen occurs from solids (phytomass, necromass) that are also the sites of microbial attachment. Thus, ammonification and nitrification are presumed to be occurring at exactly the locations where TKN is generated, and no mass transfer step is involved to slow the processes. Further, agronomic ammonia application rates do not imply large oxygen transfer rates, and atmospheric reaeration is probably adequate in almost all cases. In their review of nitrogen transformations in flooded soils, Reddy and Patrick (1984) summarized estimated first-order nitrification measurements from soil studies. The first-order, volume-based nitrification rates they summarized ranged from 0.003 to 3.1 d^{-1}, with a mean of 0.29 d^{-1}. Assuming an effective nitrification depth of 30 cm in a typical wetland treatment system, these values are equivalent to area-based first-order nitrification rate constants between 0.33 and 339 m/yr with an average of 32 m/yr. It is likely that these values represent typical nitrification rate constants in systems that are not limited by a shortage of oxygen.

Depth Effects

Ammonia removal is controlled by processes that are apportioned to FWS wetland surface area. Plant biomass is an areal parameter of the wetland, for both floating and rooted species, and hence cycling is area-dependent. Volatilization, if any, is water surface-specific. Microbial attachment sites are associated with root mats, sediment–water interfaces, and litter layers. They are also associated with plant stems and leaves, which may be distributed throughout the water column. It is therefore expected that increasing water depths will not provide proportionally more removal activity.

Figure 9.43 illustrates this effect for the side-by-side tests at Arcata, California (Gearheart *et al*., 1983). Values of k_{V20} are inversely proportional to depth, meaning that doubling the depth halves the k_V-value. Then according to the definitions, the areal rate constant does not depend on depth. If the volumetric model is utilized in FWS calculations, there appears to be the option of increasing performance by increasing the water depth, and hence increasing the nominal detention time. However, that advantage is lost if the volumetric rate "constant" decreases with increasing depth, as indicated in Figure 9.43.

Ammonia Rate Constants for HSSF Wetlands

It should be noted that organic nitrogen is an important source of ammonia due to mineralization (ammonification). The HSSF wetland is typically an oxygen-limited system, and ammonification of organic nitrogen can lead to increases in ammonia concentrations within the wetland bed (Figure 9.44).

HSSF wetlands are most commonly used for secondary treatment, with a primary treatment device such as an Imhoff tank (Imhoff and Fair, 1929) or a septic tank (U.S. EPA, 1980) upstream of the wetland. HSSF wetlands are typically chosen because (if properly designed and operated), wastewater is not exposed during the treatment process, minimizing potential public health issues. Devices such as

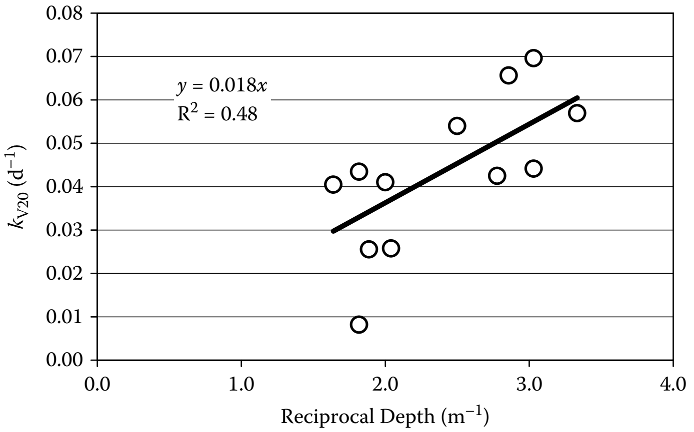


FIGURE 9.43 The dependence of volumetric ammonia rate "constants" on reciprocal depth for the Arcata, California, pilot FWS wetlands. Deeper water results in lower volumetric rate constants. (Data from Gearheart *et al.* (1983) *City of Arcata Marsh Pilot Project, effluent quality results—system design and management.* Final Report to the North Coast Regional Water Quality Board and State Water Resources Board.)

Imhoff tanks or septic tanks have anaerobic environments. In these primary treatment devices, there is usually a partial conversion of organic nitrogen to ammonia (ammonification) plus the ammonia originally present in the wastewater influent. As a result, HSSF wetlands that are designed for secondary treatment typically receive their influent nitrogen in the form of ammonia and organic nitrogen. Although the organic nitrogen fraction cannot be ignored in these cases, ammonification often does not interfere with a monotonic decreasing trend in ammonia along the flow direction (Figure 9.45).

Most HSSF wetlands are dominated by microbial processing of ammonia (Figure 9.38), and the various alternative mechanisms typically require oxygen. In passive, steady flow HSSF wetlands with no mechanical aeration, the required oxygen supply is from the air, because wetland influents typical are devoid of dissolved oxygen. The routes by which oxygen can gain entry to the water column are: (1) atmospheric diffusion and (2) plant oxygen flows (see Figure 9.40).

In the former case, oxygen diffuses from air into the water in the top bed layer. Ammonia must diffuse upward into this layer so that microbes can have both ingredients to perform nitrification. In the latter case, the oxygenated zone consists of micro zones near roots, with oxygen supplied through the plant roots. The amount of this plant aeration flux is contingent upon the root density and volume, as well as plant physiology, which dictates how much oxygen can be delivered to protect the roots from anaerobiasis. If the amount of ammonia arriving is not overwhelming, both mechanisms offer the ability to supply nitrifiers and other N-processing bacteria.

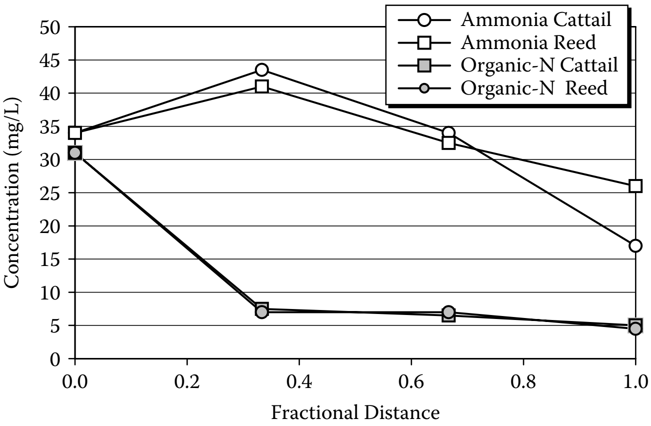


FIGURE 9.44 Profiles of ammonia and organic nitrogen through the HSSF wetlands at Thrace, Greece. (Data from Akratos and Tsihrintzis (2007) *Ecological Engineering*, 29(2): 173–191.)

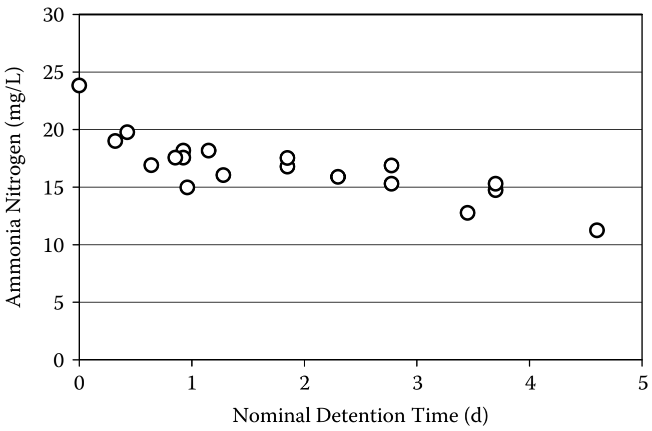


FIGURE 9.45 Decrease of ammonia along profiles through the three cells of the Minoa, New York, HSSF wetland on February 15, 1996. The cells had different hydraulic loadings, and measurements were taken at the quarter points of the flow path. (Data from Theis and Young (2000) *Subsurface flow wetland for wastewater treatment at Minoa*. Final Report to the New York State Energy Research and Development Authority, Albany, New York.)

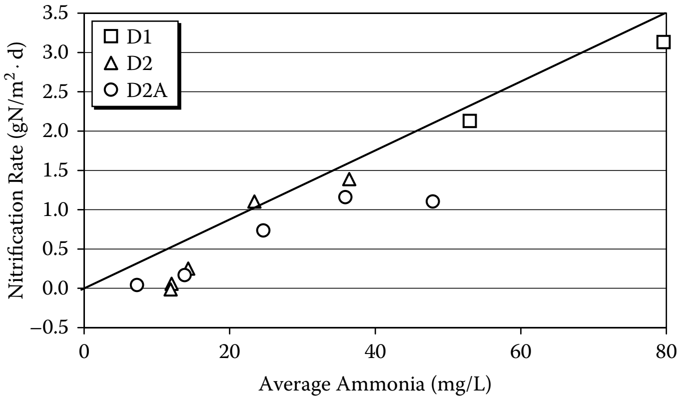


FIGURE 9.46 Dependence of nitrification rate on ammonia concentration for dairy effluents for HSSF wetlands in New Zealand. The range of inlet ammonia is restricted to C_i < 80 mg/L. For higher inlet concentrations, the apparent proportionality does not hold. (Data from Tanner *et al.* (2002a) *Ecological Engineering*, 18(4): 499–520.)

The usual model for the ammonia diffusion is a mass transfer coefficient times a concentration driving force (see Chapter 6), and thus an ammonia concentration effect is anticipated. Accordingly, a first-order areal model is expected to be operative. Rate constants are expected to be mass transfer coefficients, and thus depend upon water velocity, but only mildly upon temperature. There is ample precedent for mass transfer control of gas–liquid reactions in more controlled environments (Danckwerts, 1970).

A test of the idea of first-order removal is available from the data of Tanner *et al.* (2002a). Wetlands were operated in sequence, with each train of cells receiving a different strength of dairy wastewater. As a check of the first-order idea, Equation 9.50 is used as the basis for a plot of data. The rate of nitrification is found to be linear in the ammonia concentration for these wetland mesocosms (Figure 9.46, k = 13 m/yr, R^2 = 0.89). However, as the ammonia concentration extends to high values, in excess of 100 mg/L, the first-order concept no longer is evidenced by the data. At higher concentrations, the oxygen supply rate becomes limiting, rather than ammonia mass transfer to the reaction zone.

It is possible that, under conditions of heavy loading, the ability of the plants to defend against anoxia at their roots can be exceeded by the ammonia and BOD supply. Under that condition, plants may reduce their active root mass so that they can maintain oxic conditions near those that remain. Likewise, physical aeration may be limited by the air-side mass transfer process, meaning that all the ammonia diffusing to the surface will not be consumed.

Confirmation of these concepts is found in the data from several studies that have focused upon vertical profiles of redox, ammonia, and oxidation products (oxidized nitrogen species). The top of the bed is typically at higher redox potential, contains less ammonia, and contains more oxidized nitrogen (Figure 9.47). Similar results were found at Minoa, New York (Theis and Young, 2000), Richmond, New South Wales (Bavor *et al.*, 1988), and West Harwood, Scotland (McGechan *et al.*, 2005b).

Ranges in annual average k-rates for HSSF wetlands are summarized in Table 9.25, with the assumptions of PTIS = 6 and C^* = 0 mg/L. The median k-rate is 11.4 m/yr. However, there is wide variability in the data, and the 10th–90th percentile range is 0.4–63.3 m/yr.

Temperature dependence (θ) is summarized based on the period of record for 18 HSSF wetlands in Table 9.32. The overall data set shows little dependence on temperature (median θ = 1.014). However, the reader is cautioned that the range in θ-values is wide, and that individual systems demonstrated to have a strong dependence on temperature. As a result, use of a median θ-factor may not be appropriate if regulatory compliance for ammonia is a project objective. In Table 9.32, the 10th–90th percentile range is 0.976–1.082.

Effect of Media Size on Ammonia Removal

Based on the data that is currently available, it appears that ammonia removal is reduced as the size of the bed media increases. There are several factors that could contribute to these observations: (1) coarse bed media has less surface area (per unit volume) than finer materials, and hence less opportunity for ammonia-oxidizing biofilms; and (2) plant root growth and root penetration are inhibited in coarse bed materials (Greenway, 2002). Effect of media size on ammonia removal rates is summarized in Table 9.33.

Mulch Effects

In cold-climate applications, mulch is often used to insulate HSSF wetlands (Henneck *et al.*, 2001; Wallace *et al.*, 2001) as a means to close the energy balance without freezing the wetland (Kadlec *et al.*, 2003; Wallace and Nivala, 2005). The mulch layer reduces atmospheric exchanges, including the diffusion of oxygen.

This mulch material can degrade and exert an additional nitrogen loading on the system (Wallace *et al.*, 2001). Organic nitrogen leached from the mulch layer undergoes ammonification in the water column, imposing an additional ammonia loading on the system. The results of this mulch

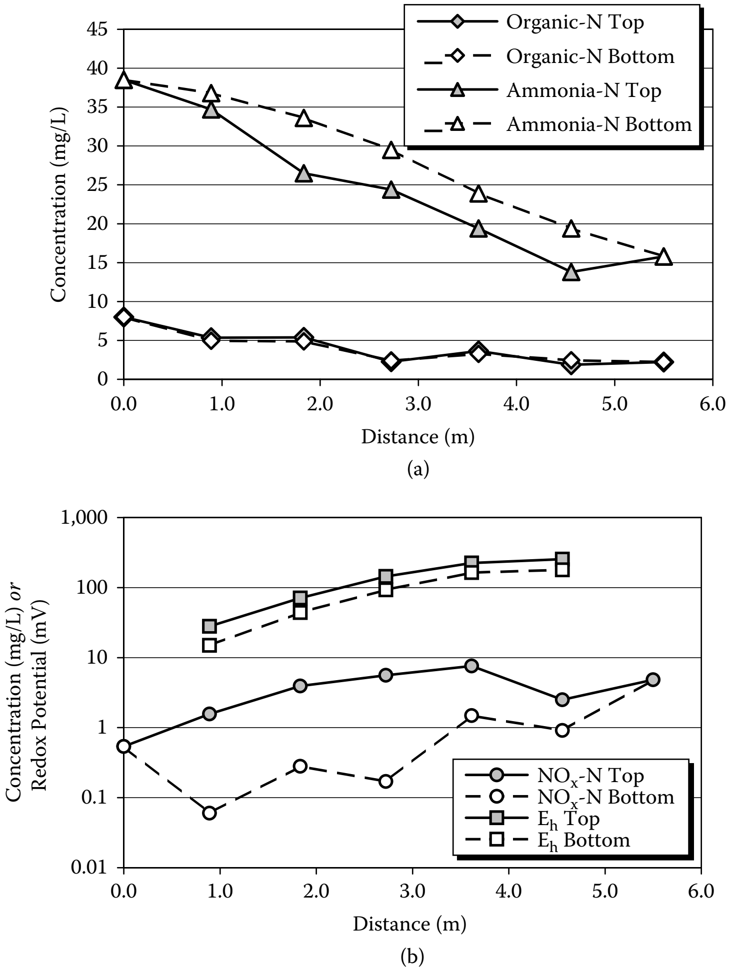


FIGURE 9.47 Longitudinal profiles in the Lismore, Australia, HSSF wetland. Samples were taken at the top and bottom of the gravel bed. (Data from Bayley *et al.* (2003) *Water Science Technology*, 48(5): 175–182.)

loading are exacerbated under situations where the mulch is a poorly decomposed material (Wallace and Knight, 2006) and the wetland experiences low hydraulic loadings. Under these conditions, the system is unlikely to respond with a *k*-rate as good as the values summarized in Table 9.25, at least for the first few years of operation, based on 21 system-years of start-up data (Wallace *et al.*, 2001).

Intrasystem Variability

Ammonia nitrogen is very frequently the target of regulatory limits. Those limits properly seek to avoid the consequences of eutrophication and possible aquatic toxicity caused by ammonia, and may be imposed at varying frequencies of measurement. Wetland design must account for both the seasonal variability in the ammonia trends, and the stochastic variation that is superimposed upon it. The general concepts have been discussed in the section on TN, and are embodied in Equations 9.40 and 9.41. Seasonal trends may or may not follow water temperatures, since agronomic (lightly loaded) systems will be heavily influenced by plant biomass cycling.

Variability in FWS Wetlands

An example of such trends and variability is given in Figure 9.48 for ammonia nitrogen at the Columbia, Missouri, FWS wetland facility. The measurement frequency was daily on weekdays. There is a pronounced annual trend, which is well represented by a cosine function. A good deterministic model would also reproduce this trend. The highest ammonia trend concentrations occur in the winter, and this cold period therefore would control design sizing, although ammonia was a design criterion for Columbia. There is a large amount of scatter of the daily concentrations about the trend line, with values both above and below the trend, as there should be for any good data fitting procedure. These daily excursions are not in general excused from regulation. The distribution of fractional departures from the trend is nearly a normal distribution for Columbia (Figure 9.48). The high concentration end of the distribution is of regulatory interest, and it is therefore useful to look at the high percentile points. The 80th, 90th, and 95th percentiles are at daily fractional excursions of 0.23, 0.33, and 0.72 of the trend, respectively.

TABLE 9.32
Temperature Coefficients for Nitrification Rate Constants in HSSF Wetlands

Site	Reference	Cell	T Range (°C)	Mean HLR (cm/d)	Mean C_i (mg/L)	Mean C_o (mg/L)	Theta
Richmond, NSW	Bavor *et al.* (1988)	Gravel	11–24	3.8	35.19	18.78	0.997
Richmond, NSW	Bavor *et al.* (1988)	*Typha*	11–24	4.6	35.19	18.80	0.998
Richmond, NSW	Bavor *et al.* (1988)	*Schoenoplectus*	11–24	5.1	35.19	19.43	1.007
Richmond, NSW	Bavor *et al.* (1988)	Mixed A	11–24	4.6	35.19	15.29	1.016
Richmond, NSW	Bavor *et al.* (1988)	Mixed B	11–24	3.8	35.19	12.64	1.012
Grand Lake, Minnesota	Unpublished	1	1–17	1.0	50.7	40.2	1.074
NERCC, Minnesota	Unpublished	1	1–16	1.4	78.2	55.3	1.017
NERCC, Minnesota	Unpublished	2	1–16	1.4	78.2	59.3	1.023
Lincoln, Nebraska	Vanier and Dahab (1997)	*Typha, Schoenoplectus*	4–21	9.5	20.4	15.7	1.057
Middleton, United Kingdom	CWA (2006)	1	4–17	15.6	2.70	0.64	1.002
West Harwood, Scotland	McGechan *et al.* (2005b)	1	5–17	5.2	—	—	1.032
Lake Capri, Missouri	Regmi *et al.* (2003)	Non-vegetated	2–24	2.3	18.9	17.0	1.100
Lake Capri, Missouri	Regmi *et al.* (2003)	Vegetated	2–24	2.3	18.9	10.4	1.110
Hamilton, New Zealand	Tanner *et al.* (1998b)	L1	10–25	1.5	29.19	18.6	1.062
Hamilton, New Zealand	Tanner *et al.* (1998b)	L2	10–25	2.5	29.19	15.9	0.976
Hamilton, New Zealand	Tanner *et al.* (1998b)	L3	10–25	3.3	29.19	19.1	0.983
Hamilton, New Zealand	Tanner *et al.* (1998b)	L4	10–25	4.9	29.19	21.2	0.975
Hamilton, New Zealand	Tanner *et al.* (1998b)	L5	10–25	6.9	29.19	22.3	0.911

Percentile	Theta
0.05	0.965
0.10	0.976
0.20	0.989
0.30	0.998
0.40	1.006
0.50	1.014
0.60	1.018
0.70	1.031
0.80	1.060
0.90	1.082
0.95	1.102

Monthly limits are a quite common averaging period. Therefore, it is useful to examine the variability of monthly average outlet concentrations, represented by a fractional addition to the trend value (Ψ), which corresponds to a multiplier on the trend value of ($1 + \Psi$). Table 9.34 lists percentiles of the monthly ammonia Ψ-distributions for some representative FWS wetlands. These data indicate that the median of the 95th percentile is an additional 110% above the trend, and that the median of the 90th percentile is an additional 87%. To incorporate this variability into design, the wetland designer must oversize the wetland, according to the steps outlined in the section on TN. Because deterministic equations (k-rate or other) represent the central estimate of performance, failure to incorporate variability means that 50% of the anticipated concentrations will be above the calculated values. If the design calculation is set at the regulatory limit, exceedance frequencies of 50% are expected to result. The risk of exceedance is lowered by increasing wetland size to the selected percentile of the Ψ-distribution.

TABLE 9.33
Effect of Gravel Size on Nitrogen Rate Constants for HSSF Wetlands

Source	Parameter	k_A (m/yr) Coarse	Fine
García *et al.* (2003b) (deep)	Ammonia	3.4	3.9
García *et al.* (2003b) (shallow)	Ammonia	5.5	8.4
Akratos and Tsihrintzis (2007)	TKN	5.1	6.7
Akratos and Tsihrintzis (2007)	Ammonia	4.9	6.3
Akratos and Tsihrintzis (2007)	Nitrification	9.3	12.3

Variability in HSSF Wetlands

Similar to FWS wetlands, HSSF wetlands display seasonal trends. Although the central tendency of the available data set indicates minimal dependence on water temperature (Table 9.32), some HSSF wetlands are clearly temperature-dependent for reasons that are not fully understood. It is also reasonable to expect that HSSF wetlands will exhibit

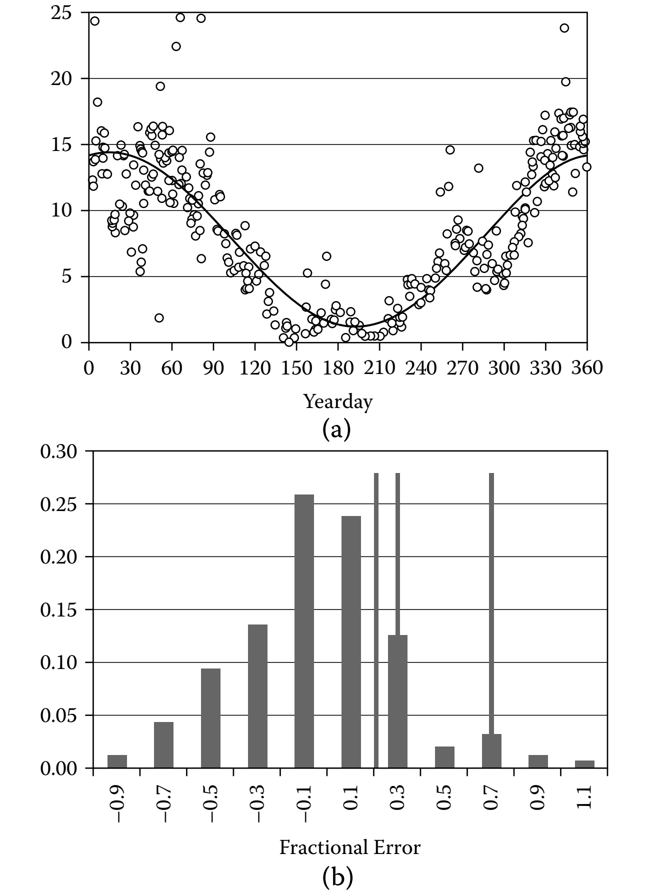


FIGURE 9.48 Distribution of daily ammonia concentrations leaving the Columbia, Missouri, FWS treatment wetland over a 20-month period (upper panel). A seasonal cosine trend is apparent, which forms the basis for computing fractional errors for each point. The distribution of errors is nearly normal (lower panel). Lines represent the 80th, 90th, and 95th percentiles at 0.23, 0.33, and 0.72, respectively.

seasonal differences due to changes in flow and loading, as well as the impact of plant biomass cycling of nitrogen (Figure 9.49).

Table 9.35 lists percentiles of the monthly ammonia Ψ-distributions for some representative HSSF wetlands. These data indicate that the median of the 95th percentile is an additional 195% above the trend, and that the median of the 90th percentile is an additional 176%. To incorporate this variability into design, the wetland designer must oversize the wetland, according to the steps outlined in the section on TN. Because deterministic equations (k-rate or other) represent the central estimate of performance, failure to incorporate variability means that 50% of the anticipated concentrations will be above the calculated values. If the design calculation is set at the regulatory limit, exceedance frequencies of 50% are expected to result. The risk of exceedance is lowered by increasing wetland size to the selected percentile of the Ψ-distribution.

TABLE 9.34
Trend Multipliers for Effluent Ammonia Nitrogen Concentrations in FWS Wetlands

System	Years of Data	Excurslon Frequency 50%	80%	90%	95%
Columbia, Missouri	3	0.97	1.37	1.70	1.80
Duplin County, North Carolina	7	0.29	1.40	1.71	2.58
Oxelösund, Sweden	5	1.00	1.29	1.38	1.41
Hassleholm, Sweden	5	0.91	1.40	1.54	1.67
Brighton, California	4	0.95	1.30	1.46	1.54
New Hanover, North Carolina	4	0.94	1.18	1.34	1.68
Linköping, Sweden	3	0.88	1.49	1.73	1.93
Musselwhite, Ontario	6	0.91	1.53	2.44	2.91
Augusta, Georgia	6	0.72	1.33	2.35	2.59
Titusville, Florida	7	0.86	1.51	2.30	2.66
Listowel 1, Ontario	4	0.95	1.53	1.91	2.10
Listowel 2, Ontario	4	0.85	1.57	1.99	2.68
Listowel 3, Ontario	4	0.96	1.56	1.91	2.73
Listowel 4, Ontario	4	0.96	1.60	1.98	2.08
Listowel 5, Ontario	4	0.89	1.45	1.65	1.80
Brawley, California	4	0.83	1.27	1.87	2.80
Imperial, California	4	0.90	1.83	2.91	3.29
Median		**0.91**	**1.45**	**1.87**	**2.10**
Mean		**0.88**	**1.45**	**1.89**	**2.25**

Note: Trend multiplier is (1 + Ψ); see Equation 6.61.

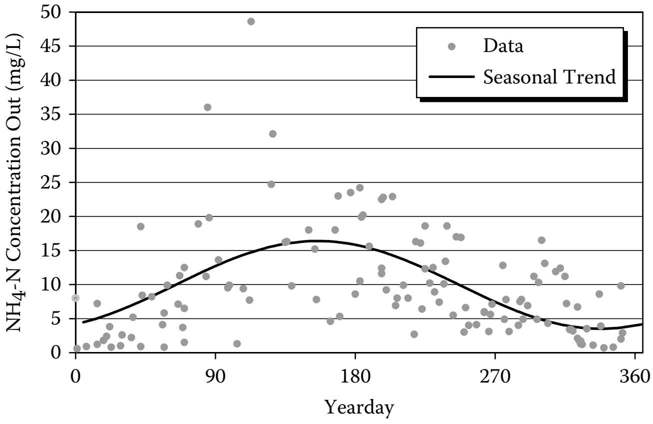


FIGURE 9.49 Seasonal changes in effluent ammonia concentrations from a HSSF wetland, Staffordshire 3, England. For this example system, a cosine trend is observed, with $A = 0.65$, $t_{max} = 156$ days, and $R^2 = 0.32$. Sampling frequency was weekly, and 2.6 years of system performance is represented. (Other systems will exhibit different seasonal changes, based on temperature, climate, latitude, and influent loadings.) (Data from CWA database (2006) *Constructed Wetlands Interactive Database, Version 9.02*. Compiled by Job and Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

TABLE 9.35
Trend Multipliers for Effluent Ammonia Concentrations in HSSF Wetlands

System	Years of Data	Excursion Frequency 80%	90%	95%	99%
Cumbria, England	9	1.12	2.40	4.79	7.84
Leicestershire 2, England	5	1.74	2.35	2.75	3.17
Ola, Arkansas	15	1.45	1.78	2.21	3.71
Dierks, Arkansas	12	1.70	2.18	2.34	2.73
Las Animas, Colorado	4	1.45	1.97	2.39	2.78
Fife, Scotland (cell 2)	3	1.14	1.26	1.29	1.41
Fife, Scotland (cell 3)	3	1.25	1.31	1.40	1.45
Fife, Scotland (cell 4)	3	1.20	1.27	1.41	1.49
Fife, Scotland (cell 1)	3	1.12	1.22	1.39	1.50
Grand Lake, Minnesota	2	1.52	2.40	2.78	3.42
Nun Monkton, U.K.	9	1.38	1.48	1.64	1.80
Waipoua, New Zealand	4	1.29	1.36	1.44	1.52
NERCC1, Minnesota	2	1.38	1.84	1.96	2.00
NERCC2, Minnesota	2	1.54	1.75	1.94	2.01
Median	**3.5**	**1.38**	**1.76**	**1.95**	**2.00**
Mean	**5.4**	**1.38**	**1.76**	**2.12**	**2.63**

Note: Numbers are the fractional multipliers (1 + Ψ) on the trend as indicated by Equation 6.61; site names for U.K. systems are approximate.

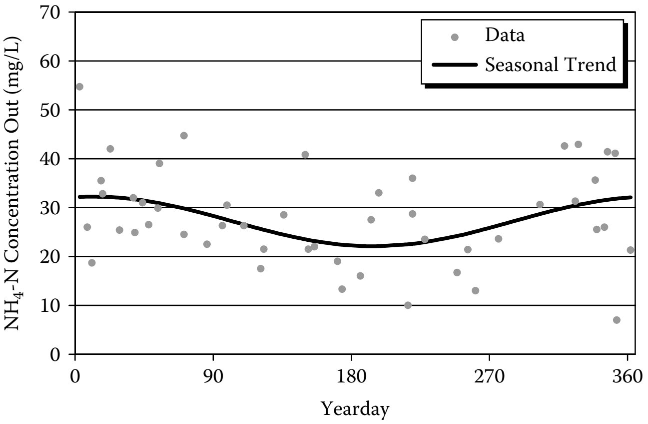


FIGURE 9.50 Seasonal changes from a VF wetland, Cornwall, England. For this example system, a weak cosine trend is observed, with $R^2 = 0.16$. (Other systems will exhibit different seasonal changes, based on temperature, climate, latitude, and influent loadings). (Data from CWA database (2006) *Constructed Wetlands Interactive Database*, *Version 9.02*. Compiled by Job and Cooper. United Kingdom Constructed Wetland Association (CWA): Gloucestershire, United Kingdom.)

Variability in VF Wetlands

While vertical flow (VF) wetlands provide an environment conducive to the oxidation of organic and ammonia nitrogen, some variability in the performance of these systems should be expected. This variance can reasonably be expected to be a combination of seasonal changes and stochastic variability. Seasonal changes in the effluent ammonia concentration for an example VF wetland system in Cornwall, England, are illustrated in Figure 9.50.

In addition to seasonal changes, VF wetlands will experience stochastic variability. Data presented over the period-of-record for the 11 pulse-loaded VF systems in Table 9.36 indicate that 10% of the time, the effluent ammonia concentration will be at least 2.27 times the mean effluent concentration.

TABLE 9.36
Trend Multipliers for Effluent Ammonia Concentrations in VF Wetlands Located in the United Kingdom

Approximate System Location	Years of Data	Excursion Frequency 80%	90%	95%
Londonderry, Northern Ireland (gravel bed)	2	1.30	1.62	1.86
Londonderry, Northern Ireland (peat bed)	6	2.51	3.17	3.58
Staffordshire 1, England (1st stage)	3	1.34	1.47	1.75
Staffordshire 1, England (2nd stage)	3	1.47	2.59	2.64
Buckinghamshire, England (1st stage)	4	3.96	5.85	7.14
Buckinghamshire, England (2nd stage)	4	2.84	4.55	11.06
Cornwall, England (1st stage)	3	1.36	1.58	1.63
Cornwall, England (2nd stage)	3	1.52	1.64	1.97
Staffordshire 2, England (1st stage)	2	1.60	2.07	2.27
Staffordshire 2, England (2nd stage)	2	1.90	2.27	2.48
Somerset, England	2	3.41	6.53	7.89
Median		**1.60**	**2.27**	**2.48**
Mean		**2.11**	**3.03**	**4.02**

Note: Data are approximately monthly. For instance, one month out of ten, we can expect an ammonia concentration 2.27 times the long-term mean value based on the median of the 11 wetlands. Trend multiplier is (1 + Ψ); see Equation 6.61. Site names for U.K. systems are approximate.

9.10 PERFORMANCE FOR OXIDIZED NITROGEN

The combination of nitrite and nitrate is oxidized nitrogen, often referred to simply as nitrate, because nitrite is usually a small fraction of the total. Nitrate can serve as a source of nitrogen for plant growth, but may not be the preferred form in the presence of ammonia nitrogen because plants must reduce nitrate prior to further use. Nitrate and nitrite are also important in water quality control because, when present in drinking water, they may result in a potentially fatal condition known as methylglobanemia, or "blue baby" syndrome because in the blood supply, the nitroso group is more readily bound to hemoglobin than oxygen. The current regulatory criteria for nitrate in groundwater and drinking water supplies in the United States is 10 mg/L. This limit has on occasion been exceeded in water supplies in the midwestern United States due to agricultural impacts, leading to deliveries of bottled water for babies and pregnant women. Eutrophication of marine environments is also a nitrate concern. The nitrogen content of the streams and rivers of the midwestern United States is of particular importance at this point in history, because of hypoxia in the Gulf of Mexico, together with the associated ecological and economic consequences (Diaz and Solow, 1999). Nitrogen pollution is of similar concern for the Baltic Sea.

Curiously, nitrate removal has been discussed in the more recent treatment wetland literature, but often with widely diverging viewpoints. U.S. EPA (1999) states that "essentially no relationship exists between nitrate loading and effluent quality …" U.S. EPA (2000a) states that the idea "Constructed wetlands can remove significant amounts of nitrogen" is a "misconception." U.S. EPA (2000a) focuses on the removal of ammonia, but indicts nitrate removal by association. In contrast, Crites and Tchobanoglous (1998) conclude that "When nitrogen is present in the nitrate form, nitrogen removal is generally rapid and complete." Crites et al. (2006) state that "nitrate will be denitrified within a few days of detention." It is the purpose of this section to present information on nitrate removal, and to set forth potential models for calculating reductions.

LOADING CONSIDERATIONS

Nitrate is potentially tied quite closely to the process of nitrification in wetlands that receive both ammonia and oxidized nitrogen, because incoming nitrate loads may be supplemented by produced nitrate. However, there may easily be confusion with processes such as ammonia volatilization (which reduces implied nitrification) and other pathways, such as the reaction of nitrite with ammonia (anammox), in which case denitrification is not necessary.

Removal of Oxidized Nitrogen in FWS Wetlands

Operational practice with FWS wetlands includes both systems that receive influent nitrogen mainly in the form of nitrate, and also systems where the influent nitrogen is present primarily as TKN and ammonia (often with significant influent BOD concentrations as well). Accordingly, a distinction is drawn here between systems receiving primarily nitrate and those receiving influential loadings of TKN, which may ultimately be dissipated by nitrification–denitrification or other alternate nitrogen chemistries.

TABLE 9.37
Annual Reduction of Nitrate Nitrogen in FWS Wetlands

Stipulations

1. Data restricted to wetlands receiving high proportions nitrate.
2. Period of record averages are used in calculations.
3. For *k*-value calculations, the following *P-k-C** parameters are selected:
   a. $C^* = 0$ mg/L
   b. P = measured *N*-value or 3 TIS
4. Ranges of variables:

	HLR (cm/d)	NO_x-N In (mg/L)	NO_x-N Out (mg/L)
Mean	11.4	12.8	5.9
Median	7.1	4.0	1.4
Max	110.0	121.0	66.0
Min	0.6	0.05	0.01

Results (N = 72 wetlands)

Percentile	Load Removed ($g/m^2 \cdot yr$)	Rate Coefficient (m/yr)
0.0	1	2.1
0.1	3	9.6
0.2	10	14.4
0.3	22	18.5
0.4	37	22.0
0.5	51	26.5
0.6	74	29.0
0.7	105	33.6
0.8	156	38.9
0.9	336	54.4
1.0	1,207	133.1

Nitrate-Rich Influents

When waters dominated by nitrate pass through a FWS treatment wetland, two processes may dissipate the oxidized nitrogen: denitrification and plant uptake. In the latter case, the nitrate used for growth is in major part returned to the ecosystem as organic or ammonia nitrogen, thus adding to the potential requirement for nitrification–denitrification.

An illustrative set of 72 nitrate-dominated FWS wetlands was analyzed for performance (Table 9.37). As for other nitrogen compounds, there is an increasing outlet concentration in response to increasing nitrate loadings (Figure 9.51). The inlet loadings span a range that includes the agronomic nitrogen loading of 120 $g/m^2 \cdot yr$, and extends well beyond. The actual nitrogen requirement of the vegetation is

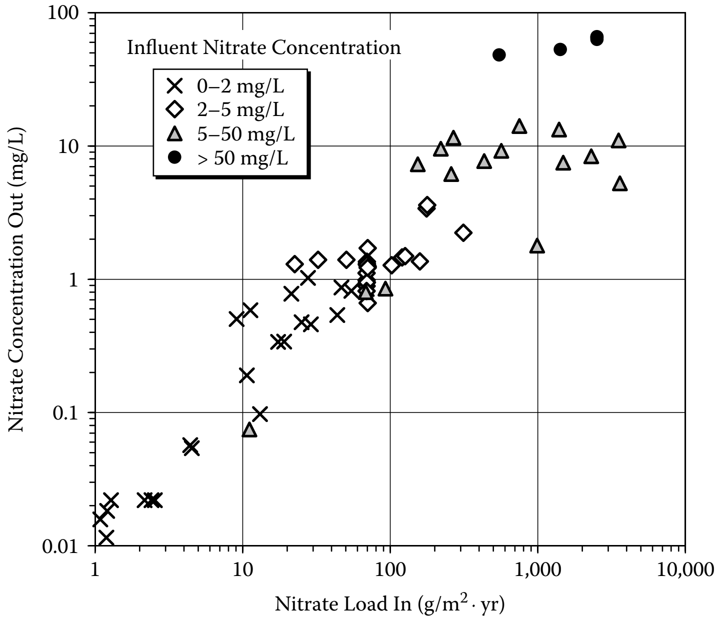


FIGURE 9.51 Nitrate outlet concentrations from FWS treatment marshes as a function of inlet loading. Each point represents the entire data set for one wetland, typically spanning one to four years. Data are from project operating records and published papers. N = 66.

less than the criterion for vegetative influence on removal, and it is doubtful that the wetland plant and algal community would derive its entire nitrogen requirement from nitrate. In general, the wetland vegetation is capable of supplying the carbon needed for traditional denitrification in these systems, although sulfur may contribute to nitrate reduction in some cases.

TKN/BOD-Rich Influents

The situation is quite different for wetlands which receive and reduce large amounts of organic and ammonia nitrogen. If sequential nitrification–denitrification is supposed, as indicated in Figures 9.14 and 9.16, then the inferred denitrification is much different from the net loss or gain of nitrate from inflow to outflow. In fact, the traditionally supposed route of full nitrification followed by denitrification sometimes becomes highly implausible based on the required oxygen and carbon supplies. To illustrate, consider the performance data for 29 FWS wetlands at four sites, used to treat animal wastewaters, which are very rich in TKN and BOD. Table 9.38 shows the annual mass balances for the various nitrogen species and their theoretical interconversions via traditional nitrification–denitrification. For these wetlands, full nitrification and BOD reduction would require a median of 32 gO/m^2·d oxygen supply, which is far above the observed reaeration potential of FWS wetlands. Further, the carbon supply for denitrification, in excess of that provided by the incoming BOD, would be a median of 7,700 g/m^2·yr of biomass decomposition, also not a realizable number (see Chapter 3). Thus, the hypothesis of traditional full nitrification followed by denitrification for these TKN-rich FWS wetlands is unlikely to be realistic, and alternate nitrogen chemistries probably play a significant role in these systems.

Alternative removal processes include anaerobic ammonia oxidation, as discussed previously, and close-coupled (simultaneous) nitrification–denitrification, in which the oxidizing power of the nitrate is recovered in ammonia oxidation. These alternatives relieve some of the need for carbon and oxygen supplies, while also eliminating the need for mass transport of nitrate to remotely located wetland anoxic zones. Whatever the mechanism(s) may prove to be, it is clear that there is not necessarily an observed large buildup of nitrate in heavily loaded wetlands.

Removal of Oxidized Nitrogen in HSSF Wetlands

Typically, HSSF wetlands have low influent nitrate concentrations, and as a result, comparison of influent and effluent nitrate values presents a very incomplete picture of nitrogen processing in these systems. Organic nitrogen may be mineralized to ammonia (via ammonification); ammonia may be oxidized to nitrite or nitrate (by conventional or alternate nitrogen pathways); and oxidized nitrogen may be reduced to N_2 or N_2O gas and expelled from the system. To gain a more complete understanding of nitrogen processing, mass balances must be used to determine the full amount of denitrification.

For 22 HSSF wetlands receiving more than 9 mg/L of oxidized nitrogen (Table 9.39), the amount of denitrification is about double the apparent nitrate removal rate. For the entire available HSSF data set of 123 wetlands, denitrification is about five times higher than the apparent nitrate removal rate (inlet–outlet). This is because many HSSF

TABLE 9.38
Examples of Theoretical Mass Balances for Traditional Nitrification–Denitrification in FWS Wetlands

System Name	Cell	NO_3–N In-Out (gN/m²·yr)	Theoretical Ammonification (gN/m²·yr)	Theoretical Nitrification (gN/m²·yr)	Theoretical Denitrification (gN/m²·yr)	BOD Removed (g/m²·d)	Oxygen Needed (g/m²·d)	Biomass Needed @15% Carbon (g/m²·yr)
Oregon State University	1	—	354	1,103	1,102	17.11	40	1,500
Oregon State University	2	—	350	1,138	1,138	18.43	42	1,260
Oregon State University	3	—	380	1,111	1,111	17.55	40	1,394
Oregon State University	4	—	326	1,028	1,028	16.33	37	1,259
Oregon State University	5	—	294	977	977	15.89	36	1,061
Oregon State University	6	—	357	1,192	1,192	18.14	42	1,757
Purdue University, Indiana	A1	−28	77	1,186	1,157	1.40	17	7,732
Purdue University, Indiana	A2	−17	74	1,253	1,236	1.42	18	8,288
Purdue University, Indiana	A3	−3	39	1,678	1,675	1.45	23	11,410
Purdue University, Indiana	A4	−8	79	1,191	1,183	1.47	17	7,888
Purdue University, Indiana	B1	−18	124	2,063	2,044	2.50	30	13,651
Purdue University, Indiana	B2	−18	110	2,199	2,180	2.53	32	14,613
Purdue University, Indiana	B3	−11	48	3,041	3,030	2.44	42	20,710
Purdue University, Indiana	B4	−25	127	1,889	1,864	2.51	28	12,364
Purdue University, Indiana	C1	−14	156	2,484	2,470	2.91	36	16,537
Purdue University, Indiana	C2	−39	88	3,254	3,215	2.57	45	21,976
Purdue University, Indiana	C3	−33	121	2,520	2,487	2.75	36	16,716
Purdue University, Indiana	C4	3	119	2,495	2,498	2.84	36	16,764
Purdue University, Indiana	D1	−50	210	4,680	4,630	5.38	67	31,025
Purdue University, Indiana	D3	5	250	4,297	4,303	4.64	61	28,965
Purdue University, Indiana	D4	−32	182	4,061	4,029	4.83	58	26,940
Pontotoc, Mississippi	1-1	0	75	391	390	0.32	5	2,663
Pontotoc, Mississippi	1-2	−2	132	504	502	0.31	7	3,464
Auburn Poultry, Alabama	1-1	7	284	616	624	3.56	13	3,126
Auburn Poultry, Alabama	1-2	−3	63	265	261	1.62	6	1,261
Auburn Poultry, Alabama	2-1	−5	293	910	904	3.13	16	5,281
Auburn Poultry, Alabama	2-2	−2	64	254	252	1.13	5	1,376
Auburn Poultry, Alabama	3-1	7	245	693	699	2.45	12	4,075
Auburn Poultry, Alabama	3-2	−16	46	239	223	2.01	6	843
Median		**−11**	**127**	**1,191**	**1,183**	**3**	**32**	**7,732**

Note: The oxygen requirement is taken as 1.5 × BOD removed + 4.6 × nitrification. The biomass carbon requirement is taken as 1.07 × denitrification.

wetlands receive and discharge little or no nitrate, but all the ammonia and organic nitrogen loss is through oxidation and reduction (regardless of the stoichiometric pathway).

Carbon Supply

It is well known that HSSF wetlands are limited in their capacity to denitrify unless an adequate carbon source is present. One source is the incoming BOD in the wastewater, and a second is the internal supply of carbon compounds due to plant biomass decomposition. For modest amounts of incoming nitrate, these two sources may be sufficient. However, if the incoming water is from pretreatment that removes BOD effectively, that source is lost. If the vegetation remains standing, biomass decomposition may occur without sending the carbon into the subsurface regions. The studies at Santee, California, showed that denitrification could be restored if carbon supplements were added to the water (methanol, molasses) or to the wetland surface (mulch) (Gersberg *et al*., 1983, 1984, 1986).

Gersberg *et al*. (1984) also showed that a top application of mulch (straw, grass clippings, marsh plant material) enhances nitrate reduction. In the absence of mulch, nitrate removal was from 9–19%. With mulch, the removal increased to 70–95%. This is presumably due to decomposition of the mulch material, which leaches organic carbon into the HSSF wetland bed. If the organic carbon mass transfer from the mulch significantly exceeds the stoichiometric demand for denitrification, BOD removal will be adversely affected (Wallace and Knight, 2006).

Removal of Oxidized Nitrogen in VF Wetlands

In general terms, vertical flow, pulse-loaded wetlands have an environment that is conducive to oxidation of organic and ammonia-nitrogen. Water is rapidly introduced into these systems to trap air within the wetland bed. Water percolates down through the wetland bed through unsaturated flow mechanisms (see Chapter 2). As a result of this flow regime, the potential for movement of atmospheric gases (notably oxygen) is much higher in VF wetlands than other types of treatment wetlands. The relative availability of oxygen in these wetlands results in a large fraction of the organic

TABLE 9.39
Annual Denitrification in HSSF Wetlands

Stipulations

1. The decomposition of 2,000 g/m²·yr of biomass causes production of 36 gN/m²·yr of organic nitrogen.
2. Inlet oxidized nitrogen above 9 mg/L.
3. Annual averages are used in calculations.
4. For *k*-value calculations, the following *P*-*k*-*C** parameters are selected:
   a. $C^* = 0.0$ mg/L
   b. $P = 8$ TIS
5. Ranges of variables:

	HLR (cm/d)	NO_x-N In (mg/L)	NO_x-N Out (mg/L)
Mean	12.9	18.7	10.0
Median	10.9	19.4	11.3
Max	41.2	36.3	25.9
Min	1.5	3.4	0.5

Results (N = 22; N·*t* = 40 wetland-years)

Percentile	Denitrification (g/m²·yr)	Rate Coefficient (m/yr)
0.05	3.3	3.3
0.10	7.4	7.4
0.20	27.8	27.7
0.30	32.7	32.0
0.40	40.7	40.1
0.50	**42.3**	**41.8**
0.60	46.9	46.4
0.70	75.4	73.0
0.80	104.9	103.2
0.90	161.5	151.8
0.95	188.9	173.2

and ammonia nitrogen being oxidized to nitrate and nitrite (Figure 9.52).

Background Concentrations of Nitrate

Nitrate is entirely consumable in treatment wetlands. The presumed value for C^* is zero for all types of treatment wetlands, because no investigation has shown a lower limit to the reduction of nitrate. Minimum reported concentrations appear to be bounded only by the analytical detection limits.

Rates and Rate Constants

In conventional activated sludge treatment system design, reduction of nitrate is often modeled with a Monod formulation, with a half-saturation constant of 0.1–0.2 mg/L (U.S. EPA, 1993b):

$$J_{AN} = \frac{k'C_{AN}}{K + C_{AN}} \tag{9.52}$$

where

C_{NN} = nitrate nitrogen concentration, mg/L (= g/m³)
J_{NN} = nitrate nitrogen removal flux, g/m²·yr
k' = Monod rate constant, m/yr
K = half-saturation constant, mg/L

Experimental studies on activated sludge indicate that the nitrate half-saturation constant (K) is in the range of 0.1 to 0.2 mg/L (U.S. EPA, 1993b). This very low half-saturation constant results in zero-order reaction kinetics (no effect of nitrate concentration) for denitrification at nitrate concentrations above about 1 to 2 mg/L. At lower concentrations, nitrate removal approaches first order. Some wetland literature presumes that nitrate reduction is zero order (Horne, 1995; White and Reddy, 2003; Mitsch *et al.*, 2005), which offers the simplistic concept of an implied constant removal rate. However, treatment wetlands are most often best described by first-order denitrification kinetics (with respect to nitrate), even at very high nitrate concentrations (Gale *et al.*, 1993; Phipps and Crumpton, 1994; Spieles and Mitsch, 2000; Hume *et al.*, 2002a; Kadlec, 2005a). Here, the first-order model is adopted, because it does the best job of describing performance in a wide variety of conditions. Batch cattail FWS mesocosm data confirms that model (Crumpton *et al.*, 1993). Full-scale wetlands also show first-order, exponential declines in nitrate, such as that observed in the Lakeland, Florida, system (Figure 9.53). The Lakeland wetland influent is nitrified, with 90% of the dissolved inorganic nitrogen (DIN) occurring as oxidized nitrogen, and therefore this system qualifies as a nitrate-dominated FWS wetland.

Therefore, the area-based first-order expression is the preferred model for the disappearance of oxidized nitrogen (denitrification and other processes combined) in wetland treatment systems:

$$J_{NN} = k_{NN}C_{NN} \tag{9.53}$$

where

k_{NN} = removal rate constant for oxidized N, m/yr

Oxidized Nitrogen Rate Constants for FWS Wetlands

Results across systems for the value $P = 3$ are given here. The value $C^* = 0.0$ mg/L is used, and the remaining model parameter is the *k*-value, selected to fit the model:

$$\frac{C_{NN,out}}{C_{NN,in}} = \left(1 + \frac{k_{NN}}{3q}\right)^{-3} \tag{9.54}$$

Seventy-two nitrate-dominated wetlands were calibrated for k_{NN}. The median annual rate constant was $k_{NN} = 26.5$ m/yr, while the average was $k_{NN} = 30.0$ m/yr (Table 9.37). The 10th–90th percentile range is 9.6–54.4 m/yr. There is a

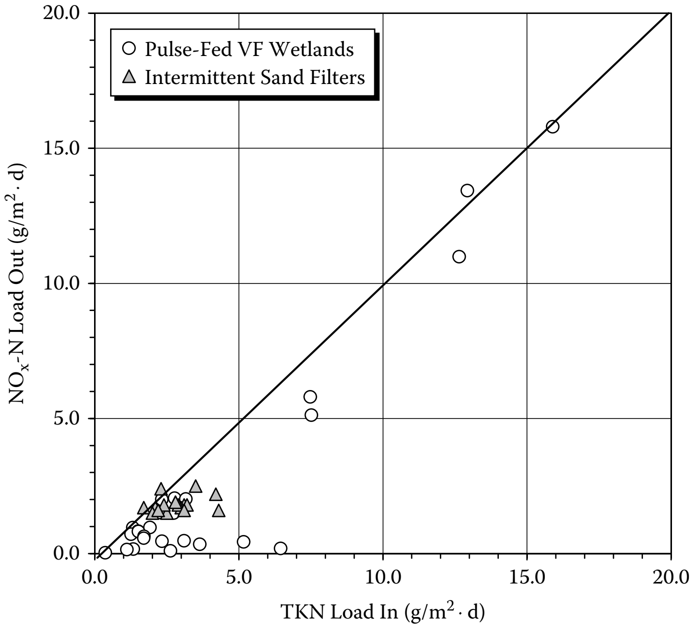


FIGURE 9.52 Export of oxidized nitrogen ($NO_3^- + NO_2^-$) in pulse-loaded vertical flow wetlands and intermittent sand filters. Data includes period-of-record performance for 20 vertical flow wetlands, annual average reductions for another 6 vertical flow wetlands (17 system-years of data), and annual average reductions for three intermittent sand filters (17 system-years of data) that were operated under similar loading regimes.

significant temperature dependence of nitrate k-values; thus even on an average annual basis, temperature or season may be an important determinant of the rate constant, and these factors are thus responsible for the some of the intersystem variability in annual k-values. Accordingly, it is necessary to examine intra-annual effects.

Results of calibration of k_{NN}-values for entire periods of record for representative wetlands are summarized in Table 9.40. Monthly averages were used to avoid synoptic error (transit time offset). Calibrations were performed for best estimates of the internal hydraulics for each wetland. Therefore, P-values range from 2 (New Hanover, measured $P = N = 2$) to near plug flow conditions, based upon system geometry. The median k_{20}-value for oxidized nitrogen is 31 m/yr, and the average is 44 m/yr.

There are strong seasonal effects on the nitrate rate constant, which is generally in synchrony with the water temperature. A temperature coefficient (θ) is capable of accounting for these effects (see Chapter 6). Temperature coefficients had a median value of 1.102, indicating a strong thermal effect on the suite of microbial processes that contribute to nitrate reduction, corresponding to a sevenfold reduction in the rate coefficient over a 20°C temperature drop.

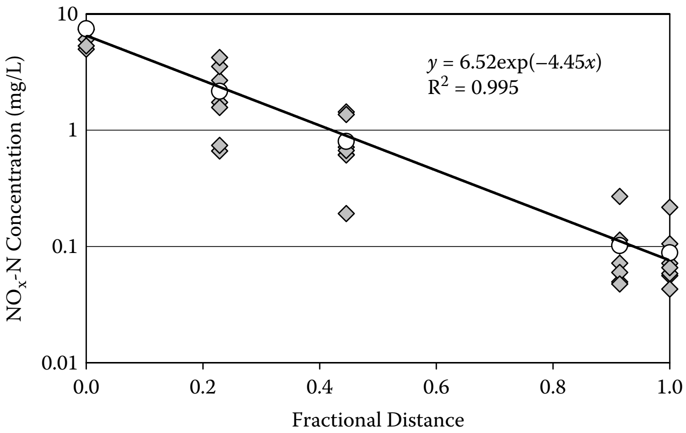


FIGURE 9.53 Oxidized nitrogen reduction in the Lakeland, Florida, FWS wetland showing first-order, exponential declines. Each point represents the annual mean performance for each of nine years of operation. (Data courtesy of the city of Lakeland.)

TABLE 9.40
Dependence of Oxidized Nitrogen Rate Constants on Temperature for FWS Systems

System	Location	Cell	HLR (cm/d)	In (mg/L)	Out (mg/L)	Load In (g/m²·yr)	*P* (TIS)	k_{20} (m/yr)	Theta	Annual T (°C)
Lakeland	Florida	Cell 1	4.46	7.58	2.44	124	3	16	1.079	22.9
Titusville	Florida	All	0.78	1.95	0.05	5	3	29	1.049	21.8
New Hanover	North Carolina	Treated	1.49	101	48	546	2	5	1.043	18.5
Eskilstuna	Sweden	Ekeby	16.43	14.25	10.62	791	6	28	1.070	8.8
Hassleholm	Sweden	Magle	16.74	12.77	10.06	212	3	24	1.091	6.8
Linköping	Sweden	All	4.38	8.59	2.29	138	3	114	1.054	7.0
Tarrant	Texas	Train 1	3.02	3.91	0.42	51	6	64	1.170	19.2
Imperial	California	All	29.3	2.27	1.69	243	∞	30	1.136	20.7
Tres Rios	Arizona	0 DZ	7.14	3.38	2.13	88	3	26	1.174	17.7
Tres Rios	Arizona	1 DZ	7.16	3.38	1.58	88	3	32	1.167	17.7
Tres Rios	Arizona	2 DZ	7.15	3.38	1.58	88	3	31	1.210	18.1
Tres Rios	Arizona	3 DZ	7.15	3.38	1.66	88	3	29	1.186	18.2
Tres Rios	Arizona	H1 Premodification	13.1	2.75	1.34	131	6	74	1.126	18.9
Tres Rios	Arizona	H2 Premodification	12.9	2.75	1.19	129	6	77	1.093	18.9
Tres Rios	Arizona	C1 Premodification	26.0	2.91	1.56	276	6	79	1.123	18.4
Tres Rios	Arizona	C2 Premodification	13.5	2.91	1.05	143	6	87	0.997	18.7
Tres Rios	Arizona	H1 Postmodification	11.4	3.77	1.14	157	6	26	1.177	19.9
Tres Rios	Arizona	H2 Postmodification	13.0	3.77	1.79	179	6	20	1.095	20.7
Tres Rios	Arizona	C1 Postmodification	65.1	4.00	3.57	950	6	55	1.059	23.9
Tres Rios	Arizona	C2 Postmodification	27.7	4.00	2.45	404	6	39	1.108	21.7
Mean								**44.3**	**1.110**	
Median								**30.6**	**1.102**	

Low BOD Wetlands

Wetlands with modest inlet BOD do not have enough carbon in the source water to drive full denitrification of the incoming nitrate load. The requirement is not precisely defined, but a range of 5–9 g BOD per gram of nitrate–nitrogen has been suggested (Crites *et al*., 2006, based upon the U.S. EPA 1993 Nitrogen Control Manual). Both nitrified municipal effluents and agricultural runoff are typically very low in BOD, and the required carbon is not present. Accordingly, traditional denitrification requires the carbon generated in wetland sediments and litter, and those are also the locations of the highest numbers of denitrifying bacteria. Nitrate must diffuse to those locations for reaction to occur.

Nitrate nitrogen is a potential nutrient source to support plant growth in the wetland. However, most waters to be treated have some ammonia, which is likely to be the preferred form for plant growth. Further, nitrate would have to diffuse through the top layers of wetland sediment to reach the root zone for uptake, which is the denitrification zone. It is therefore not surprising that there is no evidence of a seasonal nitrate uptake during the growing season. Here it is presumed that microbial processes are responsible for nitrate loss, but the pathways may include more than traditional denitrification.

In those cases where there is a significant contribution of internal wetland nitrification to the nitrate loading, a sequential correction has been made to the kinetic scheme. The method for doing so is described in more detail in Section 9.11. However, there are many situations in which there is little nitrification in the wetland, because incoming and outgoing ammonia nitrogen are low compared to nitrate. In the latter cases, the concentrations of nitrate alone suffice to estimate rate constants.

The first-order model does a creditable job of representing the behavior of this low BOD class of FWS wetlands. It provides a means of accounting for variable hydraulic loads as well as changing temperatures. Nonetheless, random variability remains, as evidenced by the model fit for the Imperial, California, wetland (Figure 9.54).

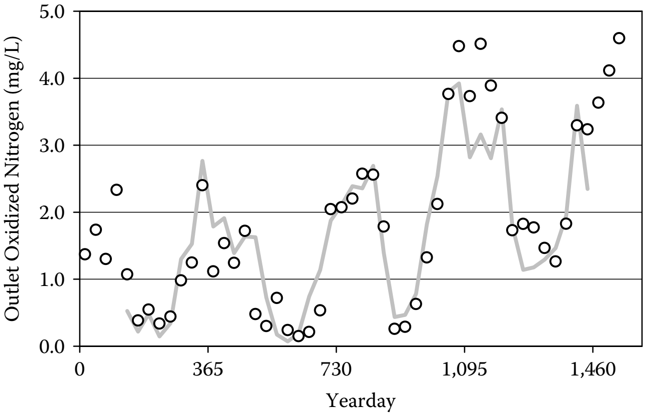


FIGURE 9.54 Model fit (line) to the monthly oxidized nitrogen data (circles) from the Imperial, California, wetland. Both temperature and flow variations are accounted.

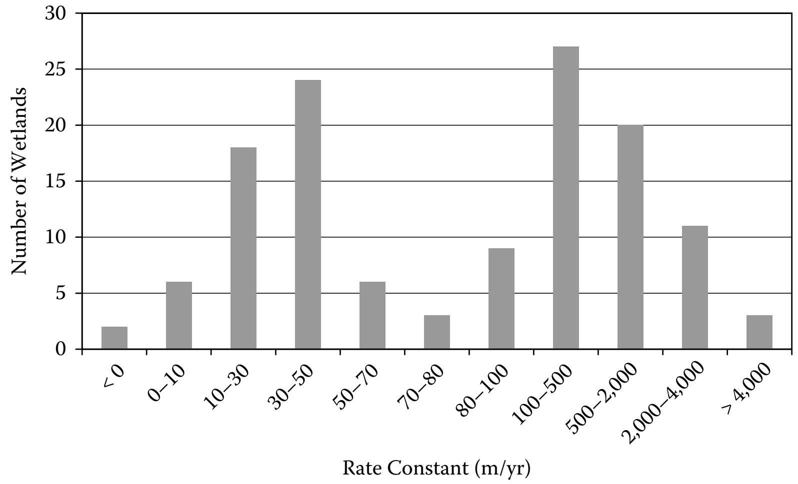


FIGURE 9.55 Numbers of FWS wetlands with 3 TIS and varying denitrification constants. Of the 129 wetlands, 43% have k_{NN} < 70 m/yr, and 53% have k_{NN} > 100 m/yr. The latter group typically have heavy BOD loads.

High BOD Wetlands

Oxidized nitrogen is processed quite differently in wetlands with high BOD, with little or no oxidized nitrogen being present in wetland waters. Wetlands in this category include those processing animal, food, and pulp and paper wastes, as well as municipal and domestic effluents high in BOD and low in nitrate. Precise criteria cannot be set from the diverse data available, but these systems possess calibrate rate constants that are 10–100 times greater than those for systems with higher nitrate and lesser BOD (Figure 9.55). For instance, pilot project studies with pretreated (partially nitrified) potato processing water showed k_{NN} = 34 m/yr for high nitrate loading, possibly under carbon limitation (Kadlec *et al.*, 1997). Subsequent full-scale operation utilized a feed-forward stream of untreated water blended with the pretreated water, and yielded k_{NN} = 370 m/yr (Burgoon, 2001).

The high k_{NN}-values in Figure 9.55 are not suitable for precise design calculations, for several reasons. Firstly, most are based upon the presumption that TKN losses result in nitrate, to the exclusion of other losses, such as volatilization or anammox-type processes. Those inferred internal nitrate loads are then presumed to be dissipated by denitrification, utilizing the first-order model assumption. The result is that large assumed denitrification must occur at low nitrate concentrations, yielding high rate constants. However, it is true that nitrate accumulations do not typically occur in these systems, and therefore small nitrate rate constants do not fit the data.

The Role of Vegetation

The temperature pattern for k_{NN} values may be confused to some degree by the annual patterns of the vegetation growth and nitrogen utilization in temperate climates. There is maximum growth in spring, and in fall a period of translocation to rhizomes. The annual uptake of nitrogen is often small but nontrivial compared to the removal rates. However, the extent to which plants use nitrate is determined in part by the availability of ammonia, which may be a preferred source of nitrogen (Martin and Reddy, 1997). Direct wetland measurements have shown uptake into shoots was relatively small—6–8% compared to that of denitrification, 61–63% (Matheson *et al.*, 2002). Weisner *et al.* (1994) measured only 6–17% of added $^{15}N–NO_3$ in plants after eight days.

One of the strongest indications that the observed nitrate reductions are microbially mediated is the rapid response of wetland ecosystems to varying inlet conditions. Time-varying nitrate concentrations follow a dynamic response that is too fast to involve the wetland vegetative uptake (Spieles and Mitsch, 2000; Werner and Kadlec, 2000b).

Unvegetated open water does not promote effective denitrification. Denitrifying microbes are overwhelmingly located on underwater solid surfaces, including the sediment–water interface and submerged plant parts and litter. Those locations with low dissolved oxygen, or low redox potential, are more effective. The data of Arheimer and Wittgren (1994) yield low calibrated *k*-values for streams (12 m/yr) and for deep water, natural "pond type wetlands" (1.5-m depth, 17 m/yr). Smith *et al.* (2000) have shown nitrate removal proportional to number of shoots in a *Schoenoplectus* spp. wetland. Therefore, vegetated systems are to be preferred.

Deeper water is of little benefit after the anoxic sites in sediments and litter have been immersed. Therefore, added detention time created by deep water is of little or no value in nitrate removal. Data from the Jackson Bottoms, Oregon, side-by-side study (SRI, 1990) indicate that volumetric rate constants decrease with increasing depth. This is evidenced by Figure 9.35 in which the TN is dominated by oxidized nitrogen (60%).

Submergent as well as emergent vegetation has the advantage of producing available carbon within the water column of the wetland. Epiphytic biofilms on living and dead plant material add to the denitrification potential of the system (Eriksson and Weisner, 1997). However, the oxygen production of living underwater plants inhibits denitrification, and therefore the balance of effects must be evaluated from field data. Weisner *et al*. (1994) found *Potamageton* to be more effective than *Glyceria*, and *Phragmites* stands to be better than open water. Eriksson and Weisner (1997) measured very high rates of denitrification in a reservoir with dense *Potamageton pectinatus*. Conversely, Gumbricht (1993a) found low rates for *Elodea canadensis*. Data from 8 of the 61 systems are in the range of $26 < k < 55$ m/yr, with an average of 37 m/yr. These included values that are characteristic of the central portion of the distribution for nonwoody emergent marshes (see Figure 9.55) with low BOD.

These considerations lead to the conclusion that fully vegetated marshes with either emergent or submergent communities are the preferred option for nitrate reduction. Weisner *et al*. (1994) also reached this conclusion, and suggested that an alternating banded pattern perpendicular to flow would additionally provide hydraulic benefits.

Carbon-Starved Wetlands

At the opposite extreme, denitrification is sometimes observed to occur under conditions of inadequate carbon supply. Examples include the unvegetated (post) Tres Rios wetlands H1 and H2 in Phoenix, Arizona (Kadlec, 2006b), the Imperial, California, wetland (TTI and WMS, 2006), and the Irvine Ranch Water District wetlands in San Joaquin, California (Fleming-Singer and Horne, 2006), which were essentially unvegetated ponds receiving nitrate but little or no carbon. For the first two of these, $20 < k_{NN20} < 30$ m/yr (Table 9.38); for Irvine Ranch, $22 < k_{NN} < 34$ m/yr. A fourth example is the Musselwhite, Ontario, wetland, which receives ponded gold mine wastewater containing essentially no carbon (Bishay and Kadlec, 2005), but had $k_{NN20} = 170$ m/yr. In none of these cases was the vegetation adequate to support the observed nitrate losses.

There is a possibility of algal activity contributing to nitrate removal. Two processes might contribute: uptake and export of phytoplankton, and carbon generation via decomposition of algal debris, capable of supporting denitrification. The first process was examined by Fleming-Singer and Horne (2006), and estimated to be only about 20% of the loss of oxidized nitrogen. Algal productivities at San Joaquin were inadequate to provide the carbon necessary for denitrification.

At least two other possibilities exist whereby the traditional concept of carbon-fueled denitrification may be a secondary route of nitrate loss. As discussed above, sulfur-driven autotrophic denitrification is possible if there is enough sulfur in the waters to provide sulfide in anoxic sediments. That is the case for Tres Rios (approximately 200 mg/L sulfate sulfur in the water), and for Imperial (approximately 600 mg/L SO_4-S), which led Wass (2003) to suggest this route for nitrate loss. The second alternative route is anammox, or equivalent close-coupled nitrification–denitrification mechanisms. This may have been a contributing process at Musselwhite, because of the relatively large abundance of nitrite in the incoming water (Bishay and Kadlec, 2005).

Oxidized Nitrogen Rate Constants for HSSF Wetlands

Results across systems for the value $P = 8$ are given here. The value $C^* = 0.0$ mg/L is used, and the remaining model parameter is the k-value, selected to fit the model:

$$\frac{C_{NN,out}}{C_{NN,in}} = \left(1 + \frac{k_{NN}}{8q}\right)^{-8} \tag{9.55}$$

Seventy-two nitrate-dominated wetlands were calibrated for k_{NN}. The median annual rate constant was $k_{NN} = 26.5$ m/yr, while the average was $k_{NN} = 41.8$ m/yr (Table 9.39). The 10th–90th percentile range is 7.4–151.8 m/yr.

At present, there is not a sufficient data set to determine the temperature dependency of oxidized nitrogen removal independently for HSSF wetlands, mainly because there are few applications of HSSF technology to remove oxidized nitrogen. Most HSSF wetlands have little or no oxidized nitrogen in the influent, and few are monitored for temperature. However, since reduction of oxidized nitrogen is a microbially mediated process, it is likely that a temperature dependency does exist. The limited data that is available suggests that the temperature dependency is similar to FWS wetlands (Liehr *et al*., 2000). The use of θ-factors for FWS wetlands (Table 9.40) is recommended as a substitute until more information is available.

Intrasystem Variability

In common with other forms of nitrogen, oxidized nitrogen is susceptible to random variation around annual trends. Those trends are reasonably well described by P-k-C^* models, but monthly averages display scatter (see, e.g., Figure 9.54). The 95th percentile of that monthly scatter is contained below a line of about double the deterministic annual trend (median relative error = 1.025 × trend) (Table 9.41). The reader is

TABLE 9.41
Trend Multipliers for Effluent Oxidized Nitrogen Concentrations in FWS Wetlands

System	Years of Data	Excursion Frequency 50%	80%	90%	95%
Lakeland 1	4	0.79	1.25	1.47	1.64
Linköping	3	0.60	1.46	1.83	2.30
New Hanover	2	0.97	1.34	1.59	1.70
Magle	2	1.00	1.14	1.18	1.20
Imperial	4	1.03	1.38	1.71	2.26
Brawley	4	1.11	1.67	2.08	2.32
Ekeby	2	0.90	1.02	1.17	1.42
Musselwhite	6	1.01	1.30	1.45	1.66
Boggy Gut	5	0.78	1.73	2.25	2.50
Tres Rios H1	5	1.01	1.58	2.67	3.19
Median		**1.00**	**1.38**	**1.71**	**0.95**
Mean		**0.93**	**1.39**	**1.74**	**2.02**

Note: Trend multiplier is (1 + Ψ); see Equation 6.61.

again reminded that wetland design must account for both the seasonal variability in the nitrate trends, and the stochastic variation that is superimposed upon it.

9.11 MULTI-SPECIES NITROGEN MODELING

The nitrogen processing network exemplified by Figures 9.13–9.16 is assumed to consist of interconversions of nitrogen in the water accompanied by exchanges with the sediments and biomass and the atmosphere. Over the last decade, this network has become recognized as the fundamental basis for simulating nitrogen flows and conversions in treatment wetlands. There are, however, significant differences in literature reports, based upon different hydraulic assumptions, and upon differences in the functional form of transfer rates. As a basis for understanding, a simplified version is presented here.

SEQUENTIAL NITROGEN MODELS: AN ILLUSTRATION

A version of the nitrogen network is shown in Figure 9.56. This highly simplified illustrative nitrogen reaction network restricts nitrogen biomass uptake to ammonium. This is accounted by an uptake flux (J_{AU}), where this and other fluxes are in gN/m²·d. Biomass decomposition can also release organic nitrogen into the water via decomposition (J_R). The residual of necromass nitrogen is accreted or buried, at rate J_B. Incoming organic nitrogen loss is ascribed to ammonification (J_A). Ammonia is nitrified at rate J_N, thus adding to the amount of nitrate in the system. Nitrate nitrogen (oxidized nitrogen) from all sources is presumed to undergo denitrification, at rate J_D.

The hydraulic assumption for this illustration is taken to be steady, nonaugmented flow, with a flow pattern of TIS. Under more complicated assumptions, the water budgets for the TIS are required. This model has been described for a single removal reaction (see Chapter 6), but here it is necessary to extend the ideas to the network of conversions. This reaction network will be presumed to consist of zero and first-order reactions, which represent the simplest possible rate equations that may be chosen. The resulting model for the progress of concentrations as water moves through the wetland is tractable, but necessarily more complicated than for a single species. The mass balances are:

$$Q(C_{ON,in} - C_{ON,out}) = (-J_R + J_A)A_j \tag{9.56}$$

$$Q(C_{AN,in} - C_{AN,out}) = (-J_A + J_{AU} + J_N)A_j \tag{9.57}$$

$$Q(C_{NN,in} - C_{NN,out}) = (-J_N + J_D)A_j \tag{9.58}$$

$$\Delta N = J_{AU} - (J_R + J_B) \tag{9.59}$$

where

A_j = area of the jth tank, m²
J = transfer flux, gN/m²·d
Q = flow rate, m³/d
ΔN = increase in phytomass nitrogen, gN/m²·d

and the rate subscript notation is shown in Figure 9.56.

The rates for waterborne species are specified as first order, as described in the preceding sections of this chapter, according to Equations 9.39, 9.50, and 9.53. The rate constants are assumed to be temperature-dependent, thus requiring the monthly time series of water temperatures (Figure 9.57). Further for this illustration, the uptake, decomposition, and burial rates are specified as seasonally dependent. The monthly time series of phytomass nitrogen content is specified (estimated) (Figure 9.57). The ratio of burial-to-return from vegetation is set, and the burial assumed to be

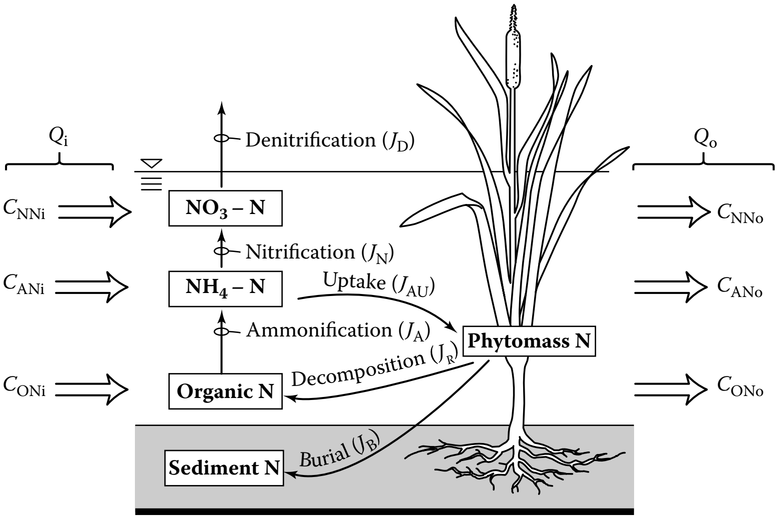


FIGURE 9.56 Conceptual model for nitrogen routing in a FWS wetland. This simplified version omits factors such as atmospheric gains and losses, infiltration, volatilization, and sorption.

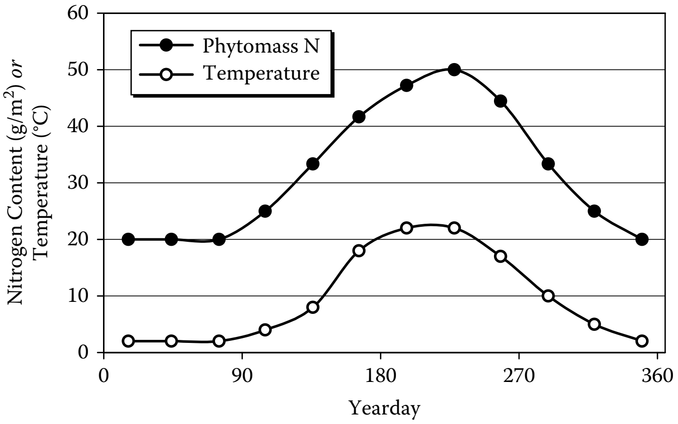


FIGURE 9.57 Time series of phytomass nitrogen and temperature for a nitrogen model illustration.

constant throughout the year. The assumed rate constants and parameters are given in Table 9.42. Results are shown in Figure 9.58.

Despite the oversimplicity of this illustration, several important features are manifest. This example is set in the range of microbial control, with an inlet ammonia loading of 548 gN/m^2·yr. First, the ammonia outlet concentrations display a midsummer minimum, and a winter maximum. However, the component nitrogen transfers are not synchronized. Plant uptake is an important contributor to the annual pattern, with a peak in the spring at maximum growth. Nitrification and denitrification peak much later, and are of the same order of magnitude as plant uptake (and release). Second, the choice of a high denitrification rate constant forces the nitrate levels to remain low throughout the year. Third, there is a considerable return flux of organic nitrogen from decomposition (peaking at 120 gN/m^2·yr during October), but the average organic nitrogen content of the water remains low (average = 1.54 mg/L).

There is no calibration data set for this illustration, but several studies have explored variants of this compartment model, and some have performed such calibrations.

TABLE 9.42
Assumed Rate Constants and Parameters for the Nitrogen Species Monthly Model Calculations

Parameter	Value	Units
End-of-season phytomass	2,500	g/m^2
	50	gN/m^2@2%
Winter phytomass	1,000	g/m^2
	20	gN/m^2@2%
Phytomass burial	10	gN/m^2·yr
Phytomass decomposition	40	gN/m^2·yr
Ammonification k_{20}	30	m/yr
Ammonification θ	1.02	—
Nitrification k_{20}	20	m/yr
Nitrification θ	1.06	—
Denitrification k_{20}	400	m/yr
Denitrification θ	1.10	—
Number of TIS	3	—
System HLR	10	cm/d
Inlet organic N	1	mg/L
Inlet ammonia N	15	mg/L
Inlet nitrate N	0	mg/L

Note: The temperature and phytomass time series are shown in Figure 9.57.

Sequential Nitrogen Models in the Literature

Nitrogen modeling may be divided into efforts targeting FWS wetlands and those which aim to simulate SSF systems. Both applications contain most of the same pieces, with SSF models typically designating a solids compartment related to the media, whereas FWS models contain one or more sediment/soil compartments. The microbial interconversions are the same in character, as is the contribution of plant growth.

Free Water Surface Systems

Dorge (1994), Jorgensen (1994)

This work considers two vertical layers: a surface layer and an active layer. Otherwise, there is no consideration of spatial variability, and it is therefore a one-tank model. The model is dynamic, utilizing a daily time-step. The waterborne species are nitrate and ammonia; water-phase organic nitrogen is not considered. Hydrology may be variable, in response to rain and evapotranspiration. Plant uptake is assumed first order, while nitrification and denitrification are described by Monod kinetics. All rate constants are considered to be temperature-dependent. The principal calibration parameters were for mineralization of detrital nitrogen, and uptake rates for nitrate and ammonia. Calibration was conducted for three event-driven Danish wetlands, with good success; but these were not constructed treatment wetlands.

Kadlec and Knight (1996)

The previous edition of this book contains a description of a steady-state model for accounting the various nitrogen

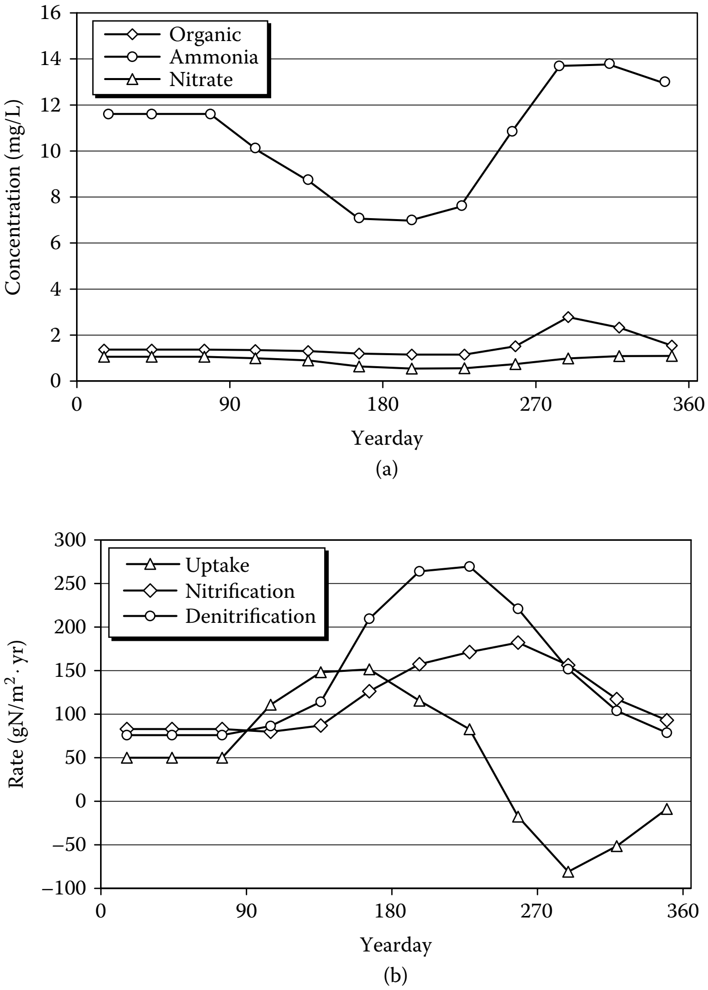


FIGURE 9.58 Illustration of the monthly time series for a 3-TIS multispecies nitrogen model. (a) Concentrations; (b) monthly fluxes. The incoming water contains 1, 15, and 0 mg/L of organic, ammonia, and nitrate respectively.

species in treatment wetlands. Equations are set forth for the interconversions of waterborne organic, ammonia, and nitrate nitrogen under plug flow circumstances. Plant uptake and return are estimated and superimposed upon the water-phase model. Both ammonia and nitrate were considered available for plant uptake, according to their relative availability. First-order rates were presumed for the water phase, and zero-order for return fluxes. Arrhenius temperature dependence was assumed for rate constants. Calibrations were conducted for several treatment wetlands, primarily FWS and for organic and ammonia nitrogen, with good success.

Martin and Reddy (*1997*)

These authors set forth a spatially explicit, 2-D steady-state model based upon unvarying flow (no rain or ET effects). The simulated nitrogen species were particulate organic, ammonia, nitrate, and vegetative. Seven vertical zones were considered, consisting of surface water, an aerobic zone under it, and five anaerobic zones below the aerobic zone. Rates were taken to be either zero or first order, with no temperature or seasonal dependence. Ammonia and nitrate were considered available for plant uptake, according to their relative availability. Soil sorption and volatilization of ammonia were included. Movement into the soil zones was attributed to diffusion, but diffusion coefficients had to be included in the calibration set, with very large alterations. Longitudinal compartmentalization was accomplished by dividing the wetland into segments of one-day detention time. Therefore, the model had variable numbers of TIS, corresponding to the number of days of detention. No calibration was attempted, but literature was used to set model parameters and explore sensitivity. Consequently, many assumptions were untested in the treatment wetland environment.

Gerke et al. (*2001*)

The FWS wetland at Kingman, Arizona, was the platform for a sequential nitrogen model that included ammonification, nitrification, and denitrification. Plug flow was assumed, because the system comprised three cells in series, each with an aspect ratio of 14. Despite the fact that the wetland was observed to have maximum ammonia removal in April, during maximum plant growth, this model had no plant uptake or release component. Rate constants for organic and ammonia did not display temperature control, but the denitrification constant was very much greater in summer than in winter. First-order areal removal kinetics were presumed, but use of Monod kinetics as tested and found to offer no improvement. Calibration and verification for data from the wetland produced reasonable results, but the authors concluded that calibration was site-specific.

Howell et al. (*2005*)

The Hemet/San Jacinto constructed wetland provided data for calibration of a TIS model including sequential nitrogen conversions. The water-phase conversions are modeled as either first order or Monod, but Monod half-saturation constants were determined to be unimportant. Plant uptake was considered minor, based upon 10% annual turnover in aboveground biomass. No temperature effects were included, despite the fact that ammonia removal was greater in summer. Large seasonal swings in ammonia removal were ascribed to algal uptake, with the summer increase attributed to light availability. The layout of the wetland is star-shaped, with flow inward from the points of unequal arms. However, for computational convenience, a single linear flow geometry, consisting of 10 TIS, was assumed. Model parameters were adjusted to provide a best fit to data. Not surprisingly, algal cycling parameters were found to be the most sensitive.

Subsurface Flow Systems

McBride and Tanner (*2000*)

This model was built upon the IAWQ model for activated sludge processing of nitrogen, and involved 11 state variables, including biomasses of autotrophic and heterotrophic organisms. Plant uptake was considered, but temperature was not. Sorption to the gravel substrate was found to be an important process. The model was calibrated to dynamic mesocosm operations, with high ammonia (approximately 80 mg/L starting). The hydraulic framework was a single well-mixed unit. The model contained 28 parameters, and was capable of good calibration fits.

Wynn and Liehr (*2001*)

This model included the nitrogen and carbon cycles, autotrophic and heterotrophic bacterial growth, and water and oxygen balances. The TIS hydraulic framework was adopted. A plant growth component was included, but substrate sorption was not. Monod bacterial kinetics were presumed, with temperature coefficients. Bacterial parameters were the most sensitive. The model was calibrated to the Mayo, Maryland, treatment wetland, which treats nitrified influent. Model predictions were not impressive.

Langergraber (*2001*)

This modeling effort simulated 12 constituents and 9 processes. Included were DO, chemical oxygen demand (COD), ammonia, nitrite, nitrate, phosphorus, and autotrophic and heterotrophic organisms. Adsorption was considered, but plant uptake was considered negligible. Volatilization was not considered. Temperature effects were modeled. Dynamic saturated and unsaturated flows were considered, in horizontal and vertical flow SSF wetlands. The hydraulic framework was plug flow plus dispersion. Calibration fits to data were generally good.

Senzia et al. (*2002b*)

This model considered organic, ammonia, and nitrate in water, together with nitrogen in plants and sediments. The computations include dynamics, but are restricted to a single well-mixed unit. Temperature-dependent Monod kinetics were used for microbial processes, with plant uptake being drawn from both nitrate and ammonia in proportion to their abundance. Calibration and validation were carried out using data from the HSSF wetland at the University of Dar es Salaam. The system loading of 530 gN/m^2·yr was reduced by net plant uptake (71 gN/m^2·yr), accretion (102 gN/m^2·yr), and nitrification–denitrification (80 gN/m^2·yr).

Liu and Dahab (*2004*)

These authors considered each of three SSF wetlands at Lincoln, Nebraska to behave as three TIS in a dynamic simulation. Water-phase concentrations of organic, ammonia, and nitrate nitrogen were assumed to follow first-order temperature-dependent kinetics. Plants were assumed to take up both nitrate and ammonia according to zero-order kinetics. The inflow of 850 gN/m^2·yr was dominated by ammonia (67%). Nitrogen losses were equally attributed to vegetative uptake and to nitrification–denitrification.

Sequential Modeling Generalizations

All the existing efforts to account for the speciation of nitrogen as water progresses through treatment wetlands have some elements in common. The water-phase species are organic, ammonia, and oxidized nitrogen. Sediments are important repositories for accreted particulate nitrogen, some of which may be recyclable to the water. There is spatial variability in at least the flow direction, and at least slow dynamics are required to account for changing input flows and concentrations. Most studies agree that microbial processes are temperature-mediated, and most agree that volatilization of ammonia is not significant in most cases.

There is no agreement on the importance of plant uptake and cycling, mostly because of different perceptions of the nitrogen loadings that the wetland has seen or will see. Langergraber (2005) considers several different scenarios, all involving very high hydraulic loadings (up to 0.6 m/d) or very high nitrogen concentrations (up to 72 mg/L). It is, therefore, not surprising that in most instances Langergraber (2005) finds plant uptake to be negligible, with the exception of grey water treatment, in which gross annual plant uptake is estimated at 30 out of an applied 65 gN/m^2·yr.

Another factor that leads to confusion about the importance of plant uptake is the length of time over which uptake occurs. In temperate climates, growth occurs over a period of only three months, resulting in very much higher instantaneous rates during that period. In general, the gross annual productivity is about 1.3 times the end-of-season standing crop, and so the uptake rate during growth is several times the annual average based on standing crop. In tropical or semitropical climates, this seasonal change in standing crop does not necessarily occur, but that does not mean that biomass turnover is slow. For instance, Davis (1994) has estimated on the order of five turnovers per year in the Everglades of South Florida.

Here it is suggested that a distinction be made between wetlands functioning in the microbial regime and those functioning in the agronomic regime. In the latter case, plant cycling cannot be ignored, particularly as it changes the concepts of timing and temperature dependence of nitrogen removal.

It seems not to be critical to describe bacterial numbers, nor to include Monod kinetics for nitrification or denitrification, because sequential models contain enough parameters to provide adequate results without that level of detail.

The rates of individual conversion processes are sufficiently different that it is not sufficient to describe mixtures by the parameter of TN. Furthermore, regulatory considerations focus on nitrate and ammonia, as well as TN, thus requiring speciated design calculations. To that end, seasonality becomes a factor, and monthly or more frequent time periods need to be analyzed. Most wetlands are not single well-mixed units, or plug flow, and consequently, spatial variability should be included. At a minimum, dynamic, spatially variable spreadsheet mass balancing should be conducted.

SUMMARY

Organic, ammonia, and oxidized nitrogen forms interconvert in the wetland environment, with the net effect of reductions in virtually all cases. In situations of low nitrogen loading, the biogeochemical cycling of nitrogen is a critical part of the processing network for continuous flow wetlands. Plant nitrogen cycling is important for a major fraction of existing operating wetlands, which leads to seasonal behavior rather than temperature-driven behavior common for microbial processing. Therefore, a distinction is drawn between agronomic wetlands and microbial wetlands, although there are obviously systems with dual control. A benchmark instantaneous growing season rate is suggested to be 120 gN/m^2·yr, below which agronomic control should be considered. In more heavily loaded wetlands, there is preponderance of microbial processes, and removals are strongly temperature modulated. Plants utilize nitrate and ammonium for growth, but the majority of the assimilated nitrogen is subsequently released during death and decay. A small amount is permanently stored as new soil and sediment, on the order of 10 gN/m^2·yr.

The descriptions of microbial processes in conventional activated sludge and attached growth wastewater treatment systems partially carry over to processing in wetland systems under microbial control. However, there are wetland phenomena that are outside those patterns. Traditional nitrification–denitrification may not apply, because of alternative microbial routes, such as anammox or autotrophic denitrification. Conditions of oxygen supply and carbon supply are sometimes outside the bounds of conventional WWTP ideas.

Several features of nitrogen processing are known with some degree of certainty:

1. The profiles of TKN and TN are typically monotonic decreasing with detention time, when the input to the wetland is above background values. The background level of organic nitrogen is approximately 1.5 mg/L in FWS wetlands, and hence that is the background for TN and TKN as well.
2. Ammonia calculations must acknowledge the ammonification process, as well as nitrification and plant uptake. The background level of ammonium nitrogen is approximately zero.
3. Nitrate rate constants divide into two groups, one that follows a slow course dictated by sequential nitrification and denitrification, and one that displays almost instantaneous removal of nitrate. The latter may be attributable to availability and spatial proximity of a carbon supply. Influent BOD is one of the most labile forms of organic carbon within the wetland, and treatment wetlands can "burn" BOD to fuel denitrification. The background level of nitrate nitrogen is approximately zero.

First-order, area-based nitrogen loss models provide a suitable method for design of wetland treatment systems in most circumstances. These have the advantage of correctly describing internal phenomena in flow through wetlands, as well as describing batch wetland operation. Studies on side-by-side wetlands confirm the effects of the principal variables of inlet concentrations and hydraulic loading rates (or the equivalent detention times). The parent mass balance equation for water movement may be adjusted to fit extreme environmental conditions of precipitation or evapotranspiration. The rate equations account for return fluxes from the wetland biomass, and thus can fit the entire range of hydraulic loadings. In parameter estimation, the sequential nature of the nitrogen transformations cannot be ignored.

Wetland treatment systems consistently reduce nitrogen concentrations for many types of wastewaters. The magnitude of these reductions depends on many factors including inflow concentrations, chemical form of the nitrogen, water temperature, season, organic carbon, and dissolved oxygen. Although these factors can be incorporated with some success into design of wetland treatment systems, variability is inevitable. Regardless of the operative mechanisms, there now exist large databases that allow for identification of process rates and rate constants. Both annual and seasonal rates have been measured for dozens of wetlands, and found to cluster, but with considerable intersystem variability.

Designers may now choose the level of risk in sizing a new wetland. Because the major nitrogen transformation mechanisms vary seasonally, conservative design must be based on specific permit limits, with different assumptions used for design to meet annual, monthly, or daily limits. Data indicates that if design is based on deterministic, seasonal equations, target outflow concentrations should be divided by a sizeable factor, of about two, as compensation for the usual maximum monthly limit versus model equations.

The information in this chapter forms the basis for design sizing procedures discussed in Part II of this book. It is clear from the considerations in this chapter that nitrogen processing is too complex to admit simple rules-of-thumb for design.