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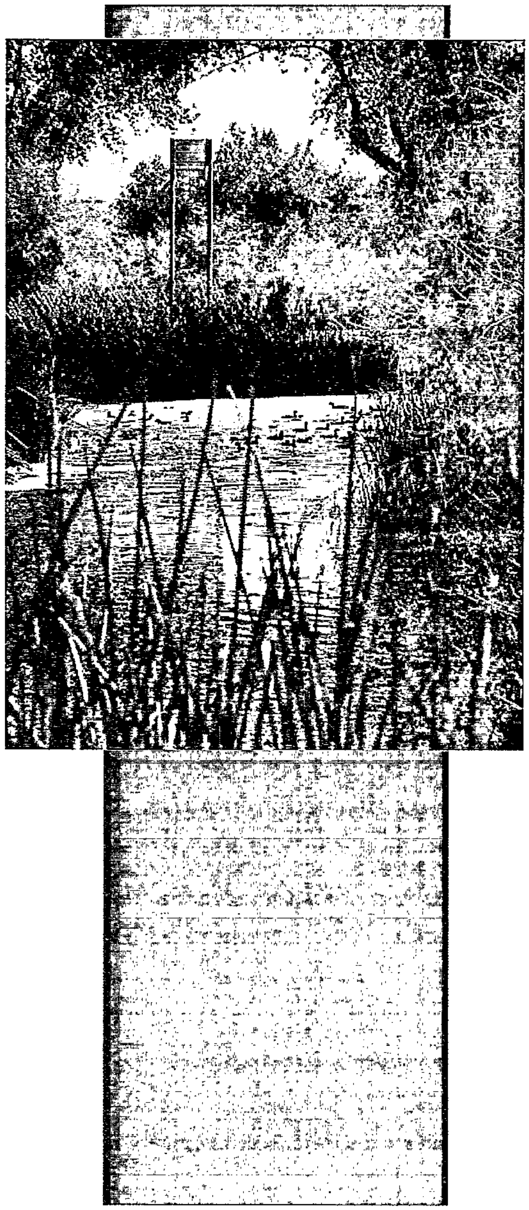
Free Water Surface Wetlands for Wastewater Treatment

A Technology Assessment

June 1999



Free Water Surface Wetlands for Wastewater Treatment:

A Technology Assessment

Prepared for

U.S. Environmental Protection Agency
Office of Wastewater Management

U.S. Department of the Interior
Bureau of Reclamation

City of Phoenix, Arizona

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Environmental Technology Initiative Program

Environmental Technology Initiative

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List of Acronyms and Symbols

ADEM	Alabama Department of Environmental Management
ADEQ	Arizona Department of Environmental Quality
ASCE	American Society of Civil Engineers
AWRA	American Water Resources Association
BOD	Biochemical oxygen demand
BOR	Bureau of Reclamation
CBOD	Carbonaceous biochemical oxygen demand
CFU/100 mL	Colony-forming units per one hundred milliliters
COE	U.S. Army Corps of Engineers
C	Centrigrade
cm	centimeter
CT	Crites, Tchobanoglous Model
d	Day
DO	Dissolved oxygen
DP	Dissolved phosphorus
EFF	Concentration reduction efficiency
ET	Evapotranspiration
F	Farenheit
FAC	Florida Administrative Code
FAP	Floating aquatic plants
FC	Fecal coliform
FWS	Free water surface
ha	Hectare
HEC2	U.S. Army Corps of Engineers computer program
HRT	Hydraulic residence time
IAW	pg. 2-3
IAWQ	International Association on Water Quality
kg	Kilogram
kg/ha·d	Kilogram per hectare per day
L	Liter
m	Meter
mm	millimeter
mg/L	Milligram per liter
min	minute
µg/L	Microgram per liter
mL	Milliliter
MPN	Maximum probable nitrogen
NADB	North American Treatment Wetland Database
NAWCC	North American Wetlands Conservation Council
NH_3-N	Ammonia nitrogen
NH_4-N	Ammonia nitrogen
NO_3-N	Nitrate nitrogen
NOD	Nitrogenous oxygen demand
NPDES	National Pollutant Discharge Elimination System
OrgN	Organic nitrogen
pH	hydrogen ion

PFR	Plug flow reactor
POP	Particulate organic phosphorus
ppb	Part per billion
ppm	Part per million
RCM	Reed, Crites, Middlebrooks Model
RED	Mass reduction efficiency
s	second
SCDHEC	South Carolina Department of Health and Environmental Control
SFWMD	South Florida Water Management District
SSF	Subsurface flow wetlands
SRCSD	Sacramento Regional County Sanitation District
SRP	Soluble reactive phosphorus
TADB	Technology Assessment Database
TIN	Total inorganic nitrogen
TKN	Total Kjeldahl nitrogen
TMDL	Total maximum daily limit
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UV	Ultraviolet
TVA	Tennessee Valley Authority
USEPA	U. S. Environmental Protection Agency
WEF	Water Environment Federation
WPCF	Water Pollution Control Federation
yr	Year

Acknowledgments

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The following individuals reviewed the multiple draft documents and contributed constructively to the final report.

Robert Bastian	-	USEPA Washington, D.C.
Robert Knight	-	Gainesville, FL
Jim Kriessl	-	USEPA Cincinnati, OH
Eric Stiles	-	Bureau of Reclamation, Denver, CO

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Data summarized in this assessment were obtained from several sources. The North American Treatment Wetland Database (NADB) report prepared by Robert Knight, Robert Kadlec, and Sherwood Reed under contract to the U.S. Environmental Protection Agency provided an initial point of entry into selecting sites to be brought up to date and for systems that met the data quality criteria. Project officers for the NADB project were Mary E. Kentula and Richard Olson at the Environmental Research Laboratory in Corvallis, Oregon, and Donald Brown at the Risk Assessment Engineering Laboratory in Cincinnati, Ohio. In addition, considerable data were obtained from owners, consultants, and researchers working on FWS constructed wetlands. Listed below are the wetland systems added to the database and individuals that provided data for this report.

Gustine, CA	Mac Walker – Larry Walker & Assoc., Davis, CA
Ouray, CO	Tom Andrews – Southwest Wetlands, Santa Fe, NM
Hemet, CA	Stella Denison – Eastern Munic. Wtr. Dist., Hemet, CA
Sacramento Regional, CA	Glen Dombeck – Nolte & Assoc., Sacramento, CA
Columbia, MO	Robert Kadlec – Wetland Mgmt. Services, Chelsea, MI
Houghton Lake, MI	Robert Kadlec – Wetland Mgmt. Services, Chelsea, MI
Minot, ND	Don Hammer – Norris, TN

Lakeland, FL	Robert Knight – CH2M-Hill, Gainesville, FL
Mount Angel, OR	John Yarnall – WesTech, Salem, OR
Arcata, CA	Robert Gearheart – Humboldt State Univ., Arcata, CA
Phoenix, AZ	Roland Wass – City of Phoenix, AZ
West Jackson Co., MI	Bill Rackley – MGCRWA, MI
Manila, CA	Wiley Buck – Manila Community Serv. Dist., Manila, CA
Beaumont, TX	Bill Benner – Beaumont, TX

The principal authors of the final report were Robert Gearheart, Brad Finney, Margaret Lang, Jeffrey Anderson, and Sophie Lagacé of Humboldt State University, while final editing was provided by Roland Wass, PBS&J.

Not all reviewer comments were able to be resolved in this final document. Participation by any specific individual in the drafting and review process does not constitute approval or endorsement of the technical content of this report. The U.S. Environmental Protection Agency and the editors of the final report accept full responsibility for any errors or omissions in this document.

SECTION 1

Introduction to Free Water Surface Treatment Wetlands

The purpose of *Free Water Surface Wetlands for Wastewater Treatment: A Technology Assessment* is to assess the application, performance, and scientific knowledge of free water surface (FWS) wetlands to treat municipal wastewater and to meet other societal and ecological needs. This report is not intended to cover subject areas to the extent needed for actual design and operation. Rather, the objective of this assessment is to produce a document that public works engineers, consulting engineers, regulatory agency representatives, researchers and citizens can use to evaluate the feasibility of FWS treatment wetland technology. The scope of this document includes a summary of the treatment processes operating in FWS treatment wetlands, a summary and evaluation of FWS treatment wetland performance, and discussion of important issues in the planning, design, and operation of FWS treatment wetlands.

Background

Free water surface wetlands have been engineered for water quality treatment in the United States since the early 1970s. Design information and operational performance data for these systems have been accumulating since that time and has led to the rapid development of a growing collection of literature. A number of efforts have been undertaken to summarize information from diverse data sources into a collection of performance descriptions. The most complete effort to date was the development of the North American Constructed Wetland Database (NADB) funded by the U.S. Environmental Protection Agency (EPA) (Knight et al. September 1993, NADB 1993, Brown and Waterman 1994).

The next step in assessing the performance of FWS treatment wetlands was to compile the assembled data into a summary of the state of knowledge. This technology assessment report describes the current understanding of processes and the performance of FWS treatment wetlands. In addition, areas of inadequate understanding are identified. The findings of this technology assessment will be incorporated into an update (in progress) of the U.S. Environmental Protection Agency's (EPA) FWS constructed wetland design manual (EPA 1988a) and the Water Environment Federation (WEF) Manual of Practice on Natural Systems (WEF, 1999), currently in preparation. Further, in the time period since the data analysis was performed for this assessment, many additional treatment wetland systems have become operational. Some of these systems have operation and performance data that are currently being used by researchers at Humboldt State University to update and provide a web-based version of the NADB by the end of 2000.

In all, three draft technical assessment documents have been prepared. A technical review team comprised of researchers, USEPA representatives, consultants, Bureau of Reclamation representatives, Corps of Engineers representatives, and municipal representatives extensively reviewed each document. This final document, *Free Water Surface Wetlands for Wastewater Treatment: A Technology Assessment*, is a culmination of an extensive effort to create an accessible summary of the operating principles and performance expectations of FWS treatment wetlands for wastewater treatment.

Introduction to the Technology

Wastewater polishing systems utilizing wetland plants have proven to be very reliable. Wetland plants create an environment that supports a wide range of physical, chemical, and microbial processes. These processes separately and in combination remove total suspended solids (TSS), reduce the influent biochemical oxygen demand (BOD), transform nitrogen species, provide storage for metals, cycle phosphorus, and attenuate organisms of public health significance. The biogeochemical cycling of macro and micronutrients within the wetland is the framework for the treatment capability of a wetland system. Valiela et al. (1976) describe the wastewater treatment capacity of natural wetlands as follows:

> *"Wetlands seem to be better processors of wastes than estuaries and coastal waters. It might be feasible to safely dispose of effluents under carefully controlled conditions on marshlands rather than deeper coastal areas where the elimination of contaminants is not as effective and dispersal of contaminants is more likely. We would like to emphasize, however, that the wetland properties outlined above, and the consequent effects on nutrients, heavy metals, hydrocarbons, and pathogens are features of wetlands as they function naturally. They are in fact providing free waste treatment for contaminated waters already."*

Natural wetlands are ecosystems that occur in areas that are intermediate between uplands and deep-water aquatic systems. Technical and regulatory definitions of wetlands focus on the dependence of wetland ecosystems on shallow water conditions which result in saturated soils, low dissolved oxygen (DO) levels or anaerobiosis in soils, and colonization by adapted plant and animal communities (Cowardin et al. 1979, Mitsch and Gosselink 1993). The ability of wetland ecosystems to improve water quality naturally has been recognized for more than 30 years (Seidel 1964). During this period, the use of constructed wetlands has evolved from a research concept to a relatively successful, and increasingly popular, pollution control technology (Tchobanoglous 1993; Kadlec and Knight 1996; Kadlec et al 2000; EPA 1999a).

Treatment Wetland Forms and Functions

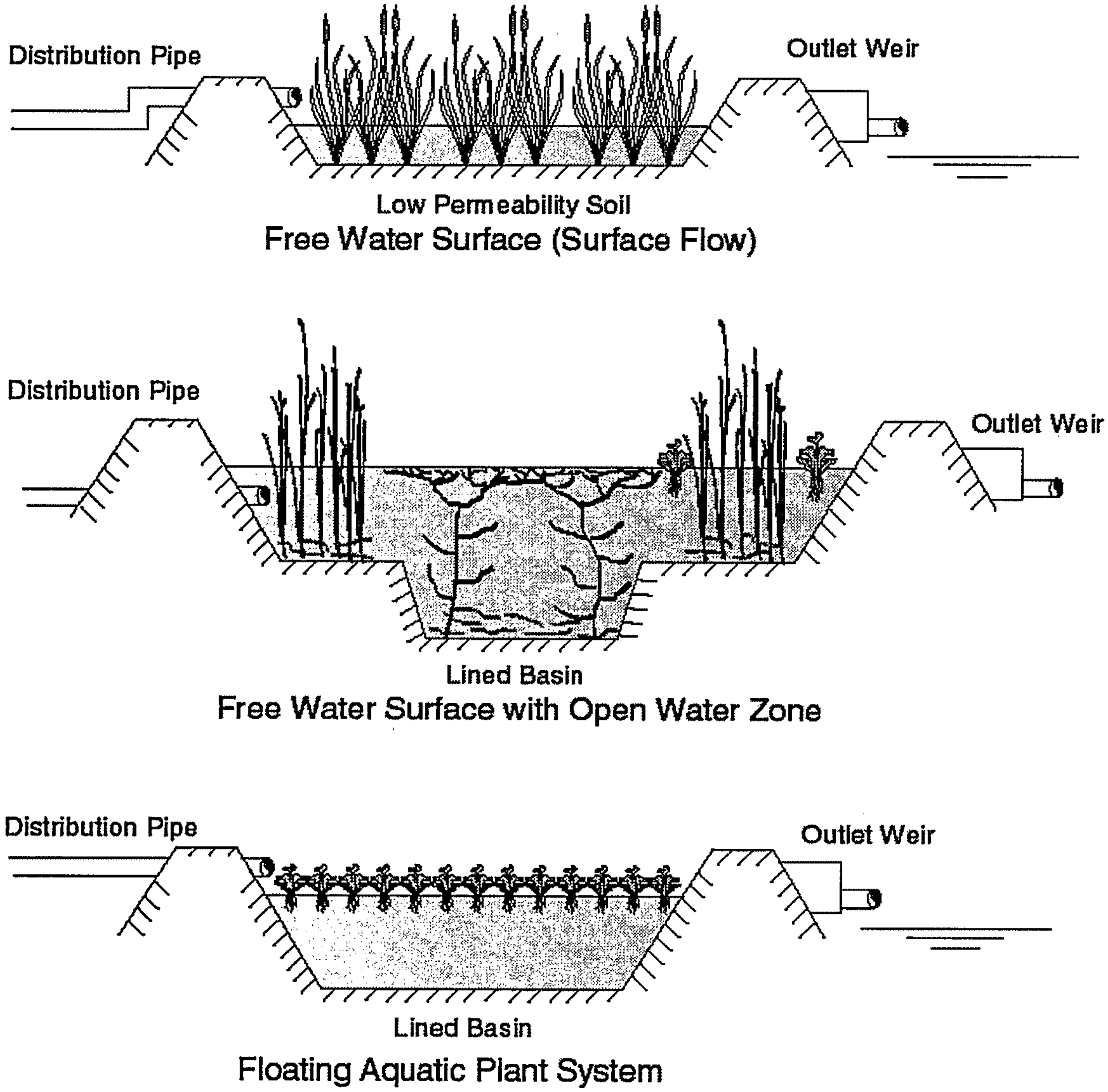
Three general types of shallow vegetated ecosystems are used for water quality treatment: (1) free water surface (FWS) wetlands, (2) subsurface flow (SSF) wetlands, and (3) floating aquatic plant (FAP) treatment systems. All three of these vegetated treatment systems are operating in the U.S. for water quality improvement. Early performance information for system types has been published in a previous design manual (EPA 1988a), and a subsurface flow technology assessment has already been completed (EPA 1993a). This technology assessment report focuses only on the FWS treatment wetland technology (Figure 1-1). In FWS treatment wetlands, water flows over the soil surface from an inlet point to an outlet point or, in rare cases, water is completely lost to evapotranspiration and infiltration within the wetland.

The technology began with the ecological engineering of natural wetlands for wastewater treatment (Ewel and Odum 1984, Kadlec and Tilton 1979). Constructed FWS wetlands are designed to mimic the hydrologic regime of natural wetlands. Currently, application of the FWS treatment wetland technology is almost exclusively through the construction of new FWS wetlands designed to meet specific influent levels and effluent water quality goals and to potentially enhance ancillary benefits associated with treatment wetland systems.

This technology assessment includes performance data from both natural and constructed free water surface wetlands. Such systems are similar in overall function with some important exceptions. The principal differences between natural and constructed treatment wetlands are structural—natural wetlands are more likely to have a forested plant community and to include a well-developed organic soil component than constructed wetlands. Natural wetlands are more likely to be subject to variable inflows and water depths and have more stagnant water zones outside the primary flow path that can reduce treatment efficiency. Also, hydraulic efficiency, the ability to utilize the entire wetland area in the process of water treatment, can be more nearly optimized in constructed wetlands than in most natural wetlands.

Free water surface treatment wetlands function as land-intensive wastewater treatment systems. Inflow water containing particulate and dissolved pollutants slows and spreads through a large area of shallow water and emergent vegetation. Particulates (typically measured as total suspended solids [TSS]) are trapped and tend to settle due to lowered flow velocities and sheltering from wind. The solids contain biodegradable organic matter, typically measured as biochemical oxygen demand (BOD) components, fixed forms of total nitrogen (TN) and total phosphorus (TP), and trace levels of metals and other recalcitrant synthetic organics. These insoluble pollutants enter the biogeochemical cycles within the water column and surface soils of the wetland. Colloidal materials are subject to flocculation and are removed partially with the particulate fraction described above. At the same time, soils and active microbial and plant populations throughout the wetland environment sorb a fraction of the dissolved BOD, TN, TP, and trace elements. These dissolved constituents also enter the overall mineral cycles of the wetland ecosystem.

FIGURE 1-1
Definition sketches for constructed wetlands: (a) free water surface constructed wetland with emergent vegetation, (b) free water surface wetland with an open water zone, and (c) constructed floating aquatic plant treatment system (adapted from Kadlec and Knight 1996).



During the process of elemental cycling within the wetland, chemical free energy is extracted by the heterotrophic biota, and fixed carbon and nitrogen are lost to the atmosphere. A smaller portion of the phosphorous and other non-volatile elements can be lost from the mineral cycle and buried in accreting sediments within the wetland. Wetlands are autotrophic ecosystems, and the additional carbon and nitrogen fixed from the atmosphere is processed simultaneously with the pollutants introduced from the wastewater source. The net effect of these complex processes is a general reduction in pollutant concentrations between the inlet and outlet of the treatment wetland.

Free water surface treatment wetlands have some properties in common with facultative lagoons, but also have many important structural and functional differences. Water column processes in the open-water zones within FWS wetlands are nearly identical to similar zones within ponds. At the surface, an autotrophic zone dominated by planktonic or filamentous algae or by floating or submerged aquatic macrophytes limits light to the deep zones. The absence of light in the deeper zones in both systems causes them to be dominated by anaerobic microbial processes. The submerged macrophytes in deep zones also afford sites for colonization of periphytic bacteria and provide substrate for algae-biofilm development.

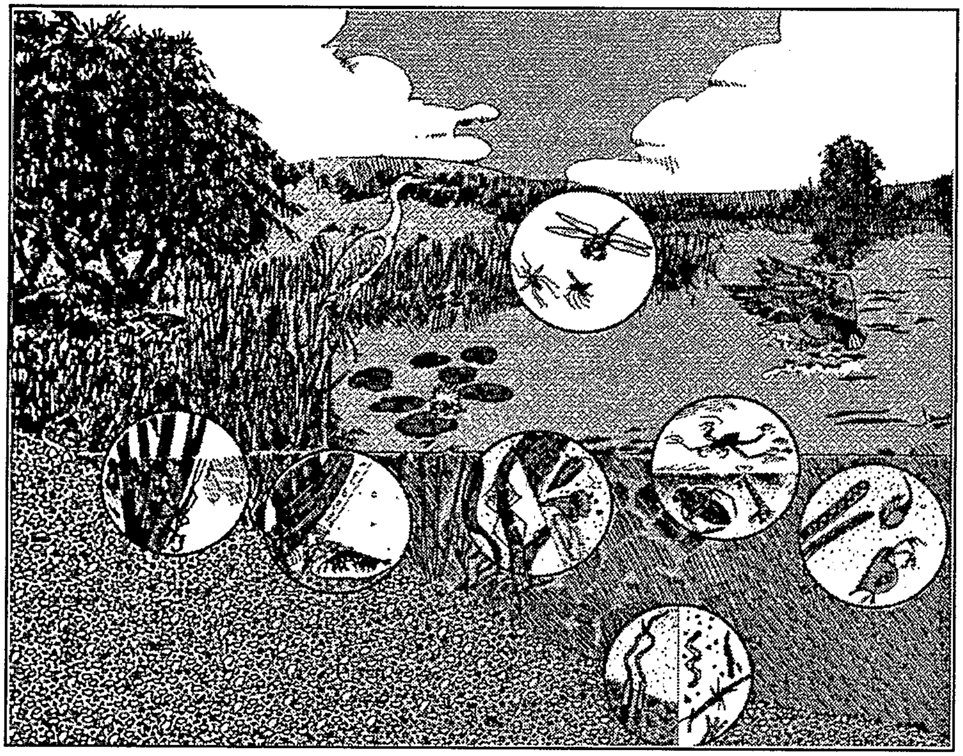
The shallow, emergent macrophyte zones present in FWS wetlands operate quite differently than any zone within a facultative lagoon. Emergent wetland plants tend to shade the water surface reducing algae growth and limiting water reaeration processes that add dissolved oxygen to the water column. Secondary populations of duckweed, covering the water surface and held in place by emergent plants, may also hinder reaeration. Net carbon production in emergent wetlands tends to be high compared to facultative ponds because of much greater primary production of plant carbon. High production of plant carbon and the resistance of plant carbon to degradation combines with a low organic carbon decomposition rate in the oxygen deficient water column to create significant differences in biogeochemical cycling rates in wetlands compared to ponds and lagoons.

Other Benefits of Treatment Wetlands

In addition to water quality benefits, wetland systems have also been designed and operated to provide wetland habitat for waterfowl and other wildlife (see Figure 1-2). Many FWS treatment wetland systems are operated as wildlife refuges or parks, as well as part of wastewater treatment, reuse, or disposal systems (Wilhem et al. 1989, Gearheart et al. 1989, Knight 1997, EPA 1999b). In some cases, FWS constructed wetland systems provide an area for public education (interpretive center or informative displays) and outdoor recreation (walking, jogging, bird watching). The design of multiple purpose use FWS constructed wetlands has been significant. More than 40 percent of the NADB secondary and 36 percent of the NADB tertiary treatment applications identified one or more additional benefits beyond that of water quality improvements. Some ecological benefits can be claimed for nearly all FWS constructed wetland systems regardless of their stated objectives. Benefits are often claimed for FWS constructed wetlands in areas where wetlands have been lost or degraded such as the facultative ponds in the north central United States (South Dakota and North Dakota) where existing degraded wetlands are used for seasonal storage.

Sometimes, the ancillary benefits of treatment wetlands work counter to those processes that improve the water quality. For example, some treatment wetlands are home, at least on a seasonal basis, for 1000s of birds and sometimes 100s of mammals, depending upon the location and scale of the system. While residing within the treatment wetlands, their activities can add bacteria and nutrients to the system. Wildlife activity may also re-suspend bottom sediments increasing turbidity and potentially causing the export of nutrients, inorganic, and organic constituents from the wetland. Wildlife induced water quality degradations are often mitigated, however, due to design factors necessary to achieve other water quality goals; e.g., providing flow time through a vegetated emergent zone prior to discharge for algal control or denitrification.

FIGURE 1-2
Ecosystem and communities of a FWS (USEPA 1993b).



Historical Development of the Technology

Treatment wetland technology using FWS wetlands has been under development, with varying success, for nearly 30 years in the United States (Table 1-1). Early laboratory studies in Germany examined the effects of emergent plants on removal of organic compounds in industrial wastewater (Seidel 1976). Constructed estuarine ponds with wetland vegetation were loaded with municipal wastewater during the 1960s and early 1970s in North Carolina (Odum 1985). Large-scale engineered natural wetland systems receiving pretreated municipal wastewater were studied in Michigan (Kadlec et al. 1993) and Florida (Ewel and Odum 1984) beginning in the early to mid-1970s. Constructed marsh-pond-meadow systems were under study at the same time in New York (Small and Wurm 1977). These research programs led to an

TABLE 1-1
Timeline of selected events in wetland treatment technology (adapted from Kadlec and Knight 1996).

Date	Location	Description
Selected Research Efforts		
1952-late 1970s	Plon, Germany	Removal of phenols and dairy wastewater treatment with bulrush plants by K. Seidel and R. Kickuth
1967-1972	Morehead City, NC	Constructed estuarine ponds and natural salt marsh studies of municipal effluent recycling by H.T. Odum and associates
1971-1975	Woods Hole, MA	Potential of natural salt marshes to remove nutrients, heavy metals, and organics was studied by I. Valiela, J.M. Teal and associates
1972-1977	Houghton Lake, MI	Natural wetland treatment of municipal wastewater by R.H. Kadlec and associates
1973-1974	Dulac, LA	Discharge of fish processing waste to a freshwater marsh by J.W. Day and coworkers
1973-1975	Seymour, WI	Pollutant removal in constructed marshes planted with bulrush by Spangler and coworkers
1973-1976	Brookhaven, NY	Meadow/marsh/pond systems by M.M. Small and associates
1973-1977	Gainesville, FL	Cypress wetlands for recycling of municipal wastewater by H.T. Odum, K. Ewel, and associates
1974-1975	Brillion, WI	Phosphorus removal in constructed and natural marsh wetlands by F.L. Spangler and associates
1975-1977	Trenton, NJ	Small enclosures in the Hamilton Marshes (freshwater tidal) were irrigated with treated sewage by Whigham and coworkers
1976-1979	Eagle Lake, IA	Assimilation of agricultural drainage and municipal wastewater nutrients in a natural marsh wetland by C.B. Davis, A.G. van der Valk, and coworkers
1976-1982	Southeast Florida	Nutrient removal in natural marsh wetlands receiving agricultural drainage waters by F.E. Davis, A.C. Federico, A.L. Goldstein, S.M. Davis, and coworkers
1979-1982	Humboldt, SK	Batch treatment of raw municipal sewage in lagoons and wetland trenches by Lakshman and coworkers
1980-1984	Listowel, Ontario	Constructed marsh wetlands were tested for treatment of municipal wastewater under a variety of design and operating conditions by Wile and associates
1979-1982	Arcata, CA	Pilot wetland treatment system for municipal wastewater treatment by Gearheart and coworkers

Date	Location	Description
1974-1988	NSTL Station, MS	Gravel-based, subsurface flow wetlands tested for recycling municipal wastewaters and priority pollutants by B.C. Wolverton and coworkers
1980 – 1989	Walt Disney World, FL	Pilot-scale wetland work on a variety of wetland plants Tom Debusk
1986	Orlando, FL	Aquatic Plants for Water Treatment and Resource Recovery by Ramesh Reddy, and Smith (1987)
1979-1998	San Diego, CA	1 mgd demonstration of treatment effectiveness of water hyacinths as a front end to the raw wastewater to potable water project
1981-1984	Santee, CA	Subsurface flow wetlands were tested for treatment of municipal wastewater by R.M. Gersberg and coworkers
1993	Hemet, CA	Effluent polishing, groundwater recharge
1994	Tres Rios, AZ	Metals removal, effluent polishing, groundwater recharge, ecosystem restoration
1994	Sacramento, CA	Metals removal, ammonia reduction, temperature reduction
Selected Full-Scale Projects		
1972	Bellaire, MI	Natural forested wetland receiving municipal wastewater
1973	Mt. View, CA	Constructed wetlands for municipal wastewater treatment
1974	Othfresen, West Germany	Full-scale reed marsh facility treating municipal wastewater in an old quarry
1975	Mandan, ND	Constructed ponds and marshes to treat runoff and pretreated process wastewater from an oil refinery
1977	Lake Buena Vista, FL	Natural forested wetland was used for year-round advanced treatment and disposal of up to 27,700 cubic meters per day (m^3/d) of municipal wastewater
1978	Houghton Lake, MI	Natural peatland receiving summer flows of municipal wastewater
1979	Drummond, WI	Sphagnum bog receiving summer flows from a facultative lagoon
1979	Show Low, AZ	Constructed wetland ponds for municipal wastewater treatment and wildlife enhancement
1984	Incline Village, NV	Constructed wetlands for total assimilation (zero discharge) of municipal effluent
1986	Arcata, CA	Constructed marsh wetlands for municipal wastewater treatment, wetland creation, and wildlife enhancement
1987	Orlando and Lakeland, FL	Two large (> 480 ha) constructed wetlands for municipal treatment
1987	Myrtle Beach, SC	Natural Carolina bay wetlands for municipal wastewater treatment
1987-1988	Benton, Hardin, and Pembroke, KY	Constructed wetlands for municipal wastewater treatment designed by the Tennessee Valley Authority
1988	Hayward, CA	Five basin 70 ha wetland for wildlife enhancement

Date	Location	Description
1988	Orange County, FL	Hybrid treatment system combining constructed and natural wetland units
1989	Sisseton, SD	102 ha total assimilation wetland treating municipal wastewater
1990	W. Jackson County, MS	Wildlife refuge linkage
1991	Columbus, MS	First full-scale constructed wetland for advanced treatment of pulp and paper mill wastewater
1991	Huron, SD	132 ha total assimilation wetland treating municipal wastewater
1991	Minot, ND	Northern surface flow wetland (51.2 ha) system for municipal treatment during 180-day discharge season
1993	Everglades, FL	Treatment of phosphorus in agricultural runoff in a 1,380-ha constructed filtering marsh
1993	Beaumont, TX	Large (263 ha) constructed marsh for municipal wastewater polishing and public use
1993	Ouray, CO	Effluent polishing
1995	Hidden Valley (Riverside), CA	Nitrogen removal, wetland restoration, wildlife habitat, groundwater recharge
1997	Cheney, WA	Wildlife enhancement, groundwater recharge

increasing number of research and full-scale treatment wetland projects treating a variety of wastewater from municipal, industrial, and agricultural sources.

Many of the earliest treatment wetlands were subsurface flow systems constructed in Europe to treat partially pretreated municipal wastewater. Soil and gravel-based subsurface flow wetlands are still the most prevalent application of this technology in Europe and the United Kingdom (Cooper 1990, Brix 1994a, EPA 1993a). Subsurface flow wetlands (SSF) using gravel substrates have also been used extensively in the United States (Reed 1992). The goal of such systems is to allow flow of polluted water through a gravel and root matrix where over time contaminants are degraded by physical, chemical, and biological processes.

Free water surface constructed and natural wetlands providing treatment beyond the secondary level were built throughout the U.S. and Canada during the 1980s and 1990s. In addition to providing advanced treatment, an increasing number of these systems have been designed and operated to enhance wildlife habitat and provide public recreation. Free water surface treatment/habitat wetlands are typically much larger than subsurface flow wetlands, including several systems greater than 400 hectares (ha) in size. In the United States, the largest application of FWS treatment wetland technology to date is the over 16,000 ha of FWS wetlands for the treatment of agricultural drainage in south Florida. Other large applications include the 89 ha wetland of Orange County, Florida, for agricultural drainage and the 1200 ha Orlando, Florida, wetland used to polish municipal effluent.

Application of the Technology

Free water surface treatment wetlands can be characterized by either their origin (natural, constructed, hybrid) or by the level of pretreatment wastewater receives prior to entering the wetland. As can be seen in Table 1-2, about 28 percent of the North American Treatment Wetland Database (NADB) treatment systems utilize natural wetlands, 69 percent of the wetlands are constructed, and 3 percent are hybrid systems. About 65 percent of the natural wetland systems are receiving conventional secondary treated wastewater. More than 45 percent of the constructed wetland systems are treating pond effluent and 22 percent are treating conventional secondary effluent. Viewed from the perspective of pretreatment levels, one-third of the wetland systems receive pond effluent, one-third receive conventional secondary effluent, and the remaining third are distributed among primary, advanced secondary, tertiary, and other.

These treatment systems were designed to meet a wide range of discharge requirements including:

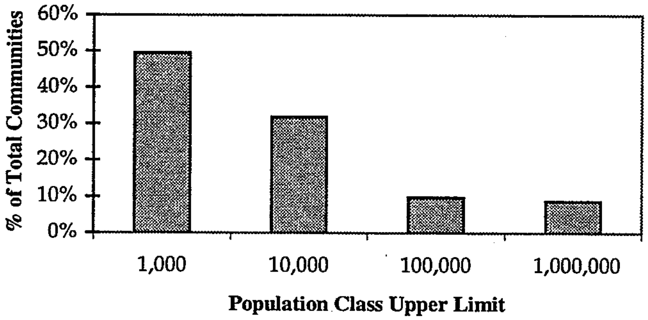
- NPDES secondary standards
- Total nitrogen
- Ammonia nitrogen
- Total phosphorus
- Total maximum daily limits (TMDL) requirements
- Advanced secondary (BOD and TSS = 10 mg/L)
- Water reuse - groundwater discharge

TABLE 1-2
Percentage distribution of NADB FWS treatment systems by wetland type and level of pretreatment.

Level of Pretreatment	Number	Natural (%)	Constructed (%)	Hybrid (%)	Other (%)	Unknown (%)
Primary	6	33	67	0	0	0
Secondary	45	53	44	0	2	0
Advance Secondary	11	18	82	0	0	0
Tertiary	4	50	25	25	0	0
Ponds	45	2	96	2	0	0
Other	4	25	75	0	0	0
None	7	43	57	0	0	0
Unknown	13	15	69	0	0	15

FWS constructed wetlands have been applied to a wide variety of community sizes, however, nearly half of the existing systems are in communities with less than 1,000 people. The fraction of systems serving (or, in the case of pilot systems, located in) communities of different populations is summarized in Figure 1-3. About 30 percent of the FWS systems have been built in communities with 1,000 to 10,000 people. There are four full-scale wetland treatment systems serving communities with populations ranging from 100,000 to 1,000,000 (Beaumont, Texas, Orlando, Florida, Hayward, California, and Riverside, California). Demonstration projects operated by Phoenix, Arizona, Albuquerque, New Mexico, and the Sacramento, California, Regional Wastewater Facility are examples of locations for potential future large community applications.

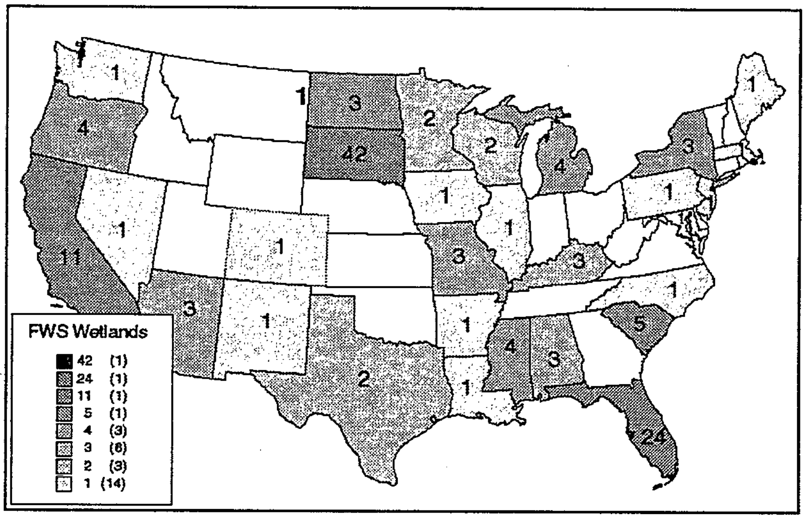
FIGURE 1-3
Percentage of all communities utilizing FWS constructed wetlands based upon community size



(NADB, n = 135).

The largest number of FWS treatment wetlands are located in the states of South Dakota and Florida (Figure 1-4). These states utilize both constructed and natural wetland systems. California has the next largest number of projects, the majority of which are designed to meet effluent polishing and water reuse objectives.

FIGURE 1-4
Distribution of FWS constructed wetlands utilized for treating wastewater by State – not including pilot projects or demonstration projects.



Summary of Technology Issues

The scope of this technology assessment is to present information that may be used to determine whether FWS wetlands are appropriate for achieving specific water quality and treatment goals. The technical tasks of primary importance for this technology include:

- Estimating accurately the influent flows and pollutant loads to the FWS treatment wetland
- Estimating wetland performance and the area and volume required to satisfy limiting water quality treatment goal(s)
- Developing wetland hydrology and hydraulic design and operating criteria to attain levels of performance comparable to the performance of the operating systems used to derive empirical rate constants
- Creating and maintaining the physical, chemical, and biological wetland system components necessary to achieve expected pollutant-processing rates

The first of these tasks, the need to predict design loading, is a standard procedure for conventional wastewater treatment technologies and is not addressed. The remaining three tasks specific to the design and operation of FWS treatment wetland technology are covered in this report.

Numerous ancillary issues are also important in the design and operation of FWS treatment wetlands, but are not covered in detail in this technology assessment. These include conventional civil engineering design criteria for dikes and levees, water inlet and outlet control structures, and soil compaction and grading; mechanical design details for flow measurement devices; and architectural/landscape design details for operator and public access.

Construction and operation issues are also important including clearing and grubbing requirements, plant selection and plant maintenance techniques, water level control, avoidance of nuisance conditions from mosquitoes or odors, operator and public safety, and wildlife management. These and other related issues for FWS treatment wetland technology are treated in greater detail in a number of sources related to FWS wetland design and operation (Mitsch & Gosselink 1993, Hammer 1996, Arizona Department of Environmental Quality [ADEQ] 1995, Kadlec and Knight 1996, Reed et al. 1995, EPA 1999a and 1988b, and Water Pollution Control Federation [WPCF, now WEF] 1989; Kadlec et al 2000).

Organization of this Report

This technology assessment report is not intended to provide detailed design guidance, but rather to present a summary of existing knowledge about FWS treatment wetland processes and performance. The goal of this report is to summarize nearly 30 years of FWS treatment wetland information. Many of the volumes documenting the development of FWS treatment wetland technology are briefly described herein and are cited in the Reference Section.

Section 2 discusses methods used to prepare this technology assessment report. Data sources are described and information concerning data quality and validation are presented. Information is also presented regarding a FWS treatment wetland technology assessment workshop convened in Mesa, Arizona, from February 2 - 4, 1996, to guide development of this report.

Section 3 summarizes key components of the physical, chemical, and biological processes that occur in FWS treatment wetlands. These fundamentals are essential for presenting and interpreting FWS wetland performance data. Subject areas covered in this section include wetland hydrology, wetland hydraulics, wetland treatment processes, wetland vegetation and vegetation patterns, and wetland thermal effects.

Section 4 presents and discusses several fundamental principles to evaluate and summarize FWS treatment wetland performance. Wetland background constituent concentrations, normal ranges of stochastic variability, and the general pattern of pollutant removal efficiencies are identified. Finally, wetland system performance is compared to regulatory permit limitations.

Section 5 identifies some system planning and design considerations. The overall goals of a FWS constructed wetland and the role they play within a watershed in terms of wildlife habitat value and water quality is examined. Environmental impact and permit issues associated with constructed wetlands are also summarized in this section. Discussed are important issues concerning wetland system planning from a community level perspective. The current FWS constructed wetland design models and methods are introduced along with a discussion of their assumptions. Finally, Section 5 includes discussion of construction, operation, and maintenance considerations, as well as monitoring and management suggestions.

Section 6 provides specific recommendations regarding the use and further development of a database for FWS constructed wetlands. Potential nuisance conditions, open water/emergent vegetation areas, major components of wetland civil design and construction, issues surrounding wildlife enhancement wetlands, multiple benefits and public access, and general operation and maintenance considerations are discussed. Last, a list of critical operational research issues is presented that if answered, would enhance the current understanding and application of FWS constructed wetlands to treat municipal and domestic wastewater flows.

SECTION 2

Methods for Technology Assessment

Technology development is an incremental process, in which initial research, guided by information from related fields, provides a preliminary assortment of observations, speculation, and conclusions. Promising observations and conclusions become the basis for the design and scope of research efforts. As a technology develops, subsequent applications can typically be categorized as those that extend the experience with the technology or those that advance the state of knowledge about the technology. In the advancement of FWS treatment wetland technology, many efforts have been made towards data compilations and feasibility assessments rather than explicit experimental studies with clear questions, replicated design, and adequate controls. The two categories are not mutually exclusive; both applications contribute to the development and acceptance of FWS treatment wetland technology, but they differ in their contribution to the advancement of the technology.

In this technology assessment, those treatment wetland applications that have been documented most thoroughly are identified and emphasized. Further preference is given to research applications designed to advance the state of knowledge about FWS treatment wetland processes and performance.

Data Sources

Information concerning FWS treatment wetlands has been published in numerous locations including agency-funded reports, wetland system design feasibility reports, system operational summaries, project case histories, technical research papers in refereed and non-refereed journals and books, conference proceedings, annotated bibliographies, design handbooks, electronic databases, and general wetland reference books. Primary sources are too numerous to include here, but citations for many of these references can be found in the documents listed in the Reference Section.

A sequence of treatment wetland conferences has been held in the U.S. and abroad beginning in the mid-1970s. A list of the major conferences and, when available, the literature citation for conference proceedings is provided in Table 2-1.

EPA has published a number of studies and summaries concerning FWS treatment wetlands. Titles and citations for these documents are summarized in Table 2-2. At least four states have published research syntheses and guidelines for consideration of treatment wetlands (Alabama Department of Environmental Management [ADEM] 1988, Arizona Department of Environmental Quality [ADEQ] 1995, Florida Administrative Code [FAC] 1989, South Carolina Department of Health and Environmental Control [SCDHEC] 1992). The

TABLE 2-1
Listing of major treatment wetland conferences.

Date	Location	Description
May 1976	Ann Arbor, MI	Freshwater Wetland and Sewage Effluent Disposal (Tilton et al. 1976)
February 1978	Tallahassee, FL	Environmental Quality Through Wetlands Utilization (Drew 1978)
November 1978	Lake Buena Vista, FL	Wetland Functions and Values (Greeson et al. 1978)
July 1979	Higgins Lake, MI	Freshwater Wetland and Sanitary Wastewater Disposal (Sutherland and Kadlec 1979)
September 1979	Davis, CA	Aquaculture Systems for Wastewater Treatment (Bastian and Reed 1979)
June 1981	St. Paul, MN	Wetland Values and Management (Richardson 1981)
June 1982	Amherst, MA	Ecological Considerations in Wetlands Treatment of Municipal Wastewaters (Godfrey et al. 1985)
July 1986	Orlando, FL	Aquatic Plants for Water Treatment and Resource Recovery (Reddy and Smith 1987)
June 1988	Chattanooga, TN	Constructed Wetlands for Wastewater Treatment (Hammer 1989)
August 1988	Arcata, CA	Wetlands for Wastewater Treatment and Resource Enhancement (Allen and Gearheart 1988)
September 1989	Tampa, FL	Wetlands: Concerns and Successes (Fisk 1989)
September 1990	Cambridge, UK	Constructed Wetlands in Water Pollution Control International Association on Water Quality (IAWQ) 2nd (Cooper and Findlater 1990)
September 1990	Show Low, AZ	Municipal Wetlands (City of Show Low Public Works Department)
June 1991	Arlington, VA	Created and Natural Wetlands in Controlling Non-Point Source Pollution (Olson 1992)
October 1991	Pensacola, FL	Constructed Wetlands for Water Quality Improvement (Moshiri 1993)
July 1992	Pinetop-Lakeside, AZ	Effluent Reuse and Constructed Wetlands (Arizona Hydrological Society Summer Seminar)
September 1992	Columbus, OH	INTECOL Wetlands Conference (Mitsch 1994)
December 1992	Sydney, Australia	Wetland Systems in Water Pollution Control IAWQ 3rd (Pilgram 1992)
November 1994	Guangzhou, China	4th International Conference on Wetland Systems for Water Pollution Control (IAWQ 1994)
April 1994	Lafayette, IN	Constructed Wetlands for Animal Waste Management (DuBowy and Reaves 1994)
July 1995	Fayetteville, AR	Animal Waste and the Land-Water Interface (Steele 1995).
September 1995	Tampa, FL	Versatility of Wetlands in the Agricultural Landscape (Campbell 1995)
May 1996	Fort Worth, TX	Constructed Wetlands for Animal Waste Management (DuBowy, in preparation)
September 1996	Vienna, Austria	5th International Conference on Wetland Systems for Water Pollution Control (Perfler and Huberl, in preparation)

TABLE 2-2
EPA Publications on Free Water Surface Treatment Wetlands.

Aquaculture Systems for Wastewater Treatment. R.K. Bastian and S.C. Reed, eds. EPA 430/9-80-006. MCD 67.
University of California, Davis – Wetland Conference Proceedings. EPA,1979.
The Effects of Wastewater Treatment Facilities on Wetlands in the Midwest. EPA 905/3-83-002. 1983.
Freshwater Wetlands for Wastewater Management. Region IV Environmental Impact Statement. Phase 1 Report. EPA 904/9-83-107. 1983.
The Ecological Impacts of Wastewater on Wetlands: An Annotated Bibliography. EPA 905/3-84-002. 1984.
Freshwater Wetlands for Wastewater Management Handbook. EPA 904/9-85-135. 1985.
Report on the Use of Wetlands for Municipal Wastewater Treatment and Disposal. EPA 430/09-88-005. 1988.
Design Manual. Constructed Wetlands and Aquatic Plant Systems for Municipal Wastewater Treatment. EPA 625/1-88/022. 1988.
Constructed Wetlands for Wastewater Treatment and Wildlife Habitat. 17 Case Studies. EPA 832-R-93-005. 1993.
Constructed Wetlands Treatment of Municipal Wastewater. Process Design Manual. EPA 625-R-99-010. Cincinnati, Ohio: Technology Transfer Branch. 1999.
Treatment Wetland Habitat and Wildlife Use Assessment Executive Summary. EPA 832-S-99-001. 1999.
Draft Guiding Principles for Constructed Treatment Wetlands: Providing for Water Quality and Wildlife Habitat. Prepared by the Interagency Workgroup on Constructed Wetlands. Available on-line at <www.epa.gov/OWOW/wetlands/constructed/guide.html>

reference section of this document also contains many detailed studies on FWS treatment wetlands.

Books dealing specifically with treatment wetlands are listed in Table 2-3. Journals that commonly publish articles about treatment wetlands are listed in Table 2-4.

Free water surface treatment wetland data summaries exist in a number of locations and include various synthesis papers (North American Wetlands Conservation Council [NAWCC] 1995, Watson et al. 1989, Water Pollution Control Federation [WPCF] 1989). The most widely used source of treatment wetland design and operational performance data is the North American Treatment Wetland Database (NADB) (Knight et al. 1993, Knight et al. September 1993, NADB 1993). This electronic database includes information from 203 treatment wetland systems at 176 sites in North America. Of these systems, 140 are FWS treatment wetlands of which 125 treat municipal wastewater, 9 treat industrial wastewater, and 6 treat stormwater.

TABLE 2-3
Books with focus on Free Water Surface Treatment Wetlands - in chronological order.

Constructed Wetlands for Wastewater Treatment: Municipal, Industrial, and Agricultural, edited by Hammer, D. A., Lewis Publishers, Michigan, 1989.
Natural Systems for Wastewater Treatment - Manual of Practice, Water Environment Federation (formally Water Pollution Control Federation), 1989.
Constructed Wetlands for Water Quality Improvement, edited by Moshiri, G. A., Lewis Publishers, Boca Raton, 1993.
Wetlands, Second Edition, by Mitsch, W. J, and J. G. Gosselink, Van Nostrand Reinhold, New York, 1993.
Natural Systems for Waste Management and Treatment, Second Edition, by Reed, S. C., R. W. Crites, and E. J. Middlebrooks, McGraw-Hill Inc., New York, 1995.
Treatment Wetlands, by Kadlec, R. H., and R. L. Knight, Lewis Publishers, Boca Raton, 1996.
Creating Freshwater Wetlands, Second Edition, by Hammer, D. A., Lewis Publishers, Boca Raton, 1996.
Small and Decentralized Wastewater Management Systems, by R.W. Crites, and George Tchobanoglous, McGraw-Hill Inc., New York, 1998
Constructed Wetlands for Pollution Control: Process, Performance, Design, and Operation, by International Water Association on Water (IWA) Specialist Group on Use of Macrophytes in Pollution Control. Scientific Technical Report No. 8, IWA Publishing, 2000, 156 pg.

TABLE 2-4
Journals that regularly publish articles dealing with treatment wetlands.

Aquatic Botany	American Water Resources Association (AWRA) Journal
Canadian Journal Fisheries and Aquatic Science	Ecological Applications
Ecological Engineering	Ecological Modeling
Hydrobiologia	Journal of Environmental Quality
Soil Science	Water Environment Research (formerly Journal of the Water Pollution Control Federation)
Environmental Science & Technology	
Water Environment Technology	Water Research
Water Resources Journal	Wetlands
Wetlands Journal	Water Science & Technology (IAWQ)

To fully evaluate the performance of full-scale FWS wetlands treating municipal wastewater, the following data and operational information (Table 2-5) should be available.

TABLE 2-5
Desired Minimum Information/Criteria for FWS Wetland Systems.

	Informational/Data Category or Criteria
1.	Municipal wastewater treatment objective with NPDES or equivalent discharge permit for target contaminants,
2.	Wetland type – constructed, natural, or hybrid,
3.	Systems have been in operation longer than 3 years and at least 2 years of operating data are available for the wetland,
4.	Spatial dimensions of the system are well characterized,
5.	Influent and effluent flow rates are available for independent wetland cells for a minimum time period of monthly averages,
6.	Influent and effluent constituent concentrations are available for independent wetland cells,
7.	Wetlands continuously discharge,
8.	Minimize use of data from leaky or infiltrating (extraneous flows in or out) wetlands,
9.	Minimize use of multiple cell wetlands without intermediate flow rate and constituent concentration data,
10.	Particulate and soluble fractionated constituent data, and
11.	Surface mapping (vegetated vs. open area) characterization is available on a regular basis.

No full-scale FWS treatment wetland system has been identified for which all of the data and operational information listed above is available. Forty FWS treatment wetland systems were judged to meet enough of conditions 1 through 6 listed above to allow adequate evaluation of the system performance. These 40 systems also include FWS treatment wetlands operating across the range of feasible pollutant loading rates. The FWS systems meeting the minimum requirements for system evaluation are listed in Table 2-6 and are the principle sources of data used for this technology assessment. For the purposes of this document, these sites will be referred to as the Technology Assessment Sites, and the data associated with these sites will be referred to as the Technology Assessment Database (TADB).

While most of these sites were represented in the NADB, several additional sites were added, and additional data from NADB sites were incorporated where available. Source information is given whenever necessary for data or information used in this report.

TABLE 2-6
FWS Wetlands used for performance evaluation (Technology Assessment Sites; Source: TADB).

System	State	Pretreatment	Seasonal	Origin	Area (ha)	Flow (m3/day)
Arcata Pilot I Cell 8	CA	Pond		Constructed	0.04	46
Arcata Pilot II	CA	Pond		Constructed	0.37	327
Arcata Treatment	CA	Pond		Constructed	1.87	6700
Arcata Enhancement Allen	CA	Pond		Constructed	4.40	5186
Arcata Enhancement	CA	Pond		Constructed	11.20	5186
Beaumont	TX	Pond		Constructed	222.00	79494
Benton Cattail	KY	Secondary		Constructed	1.50	815
Benton Woolgrass	KY	Secondary		Constructed	1.50	819
Brookhaven Meadow Marsh	NY	Primary		Constructed	0.32	48
Cannon Beach	OR	Adv Sec	x	Natural	7.00	1814
Central Slough	SC	Pond		Natural	31.60	1788
Clermont Plot H	FL	Secondary		Natural	0.20	25
Columbia	MO	Adv Primary		Constructed	38.30	54287
Fort Deposit	AL	Pond		Constructed	6.00	584
Gustine (89-90) 1A	CA	Pond		Constructed	0.39	164
Gustine (89-90) 1B	CA	Pond		Constructed	0.39	82
Gustine (89-90) 1C	CA	Pond		Constructed	0.39	41
Gustine (89-90) 1D	CA	Pond		Constructed	0.39	164
Gustine (89-90) 2A	CA	Pond		Constructed	0.39	174
Gustine (89-90) 2B	CA	Pond		Constructed	0.39	164
Gustine (89-90) 6D	CA	Pond		Constructed	0.39	144
Gustine (94-97)	CA	Pond		Constructed	9.38	2563
Houghton Lake	MI	Pond	x	Natural	75.00	4378
Iron Bridge	FL	Tertiary		Natural	494.00	45521
Lakeland	FL	Secondary		Constructed	498.00	26550
Listowel 4	ONT	Pond		Constructed	0.13	27
Manila	CA	Pond		Constructed	0.55	244
Minot	ND	Adv Sec	x	Constructed	50.18	16886
Mt. Angel	OR	Pond	x	Constructed	3.57	2320
Orange County	FL	Tertiary		Hybrid	89.00	6682
Ouray	CO	Pond		Constructed	0.89	718
Pembroke FWS 2	KY	Secondary		Constructed	0.93	287
Poinciana Boot	FL	Adv Sec		Natural	46.60	746
Reedy Creek WTS1	FL	Tertiary		Natural	35.00	12677
Reedy Creek OFWTS	FL	Tertiary		Natural	5.90	3719
Sacramento	CA	Secondary		Constructed	6.07	3975
Sea Pines Boggy Gut	SC	Adv Sec		Natural	20.00	6017
Tres Rios Hayfield	AZ	Adv Sec		Constructed	2.61	3477
Vereen Bear Bay	SC	Pond		Natural	69.00	879
West Jackson County	MS	Pond		Constructed	22.70	6257

Technology Workshop and Peer Review

A preliminary draft technical assessment document was prepared by Sherwood C. Reed in cooperation with Parsons Engineering Science, Inc., under contract with EPA. An invited workshop was convened in Mesa, Arizona, from February 2 to 4, 1996, to provide additional input to the technology assessment process. This workshop consisted of presentations and discussions of 17 FWS treatment wetland technology issues by a group of panelists who are published practitioners in this field of expertise (Table 2-7). Not all of the FWS treatment wetland professionals with valuable information could be invited to participate on this panel. However, the panel consisted of a cross-section of the types of specialists who are active in the design and operation of this technology.

These specialists brought a broad mix of experience related to different wastewater types, wetland configurations, wetland design, wetland data analysis and research, and science or engineering educational backgrounds. A revised document was prepared by Robert L. Knight and Robert H. Kadlec in cooperation with CH2M HILL, under contract with the City of Phoenix to complete tasks supported by a grant from EPA. Another revision was prepared by Robert A. Gearheart and George Tchobanoglous with extensive input from numerous reviewers. This revision reflects the data presented and discussed and insights offered by panelists at the workshop.

TABLE 2-7
Panelists for the Mesa, Arizona, workshop held February 2 through 4, 1996.

Andrews, Tom L.	Southwest Wetlands Group
Crites, Ron	Brown and Caldwell (formerly with Nolte and Associates)
DeBusk, Thomas A.	Azurea, Inc.
Dortch, Mark	U.S. Army Corps of Engineers
Gearheart, Robert A.	Humboldt State University
Hammer, Donald A.	Hammer Resources, Inc.
Kadlec, Robert H.	Wetlands Management Services
Knight, Robert L.	Private Consultant (formerly with CH2M HILL)
Mitsch, William J.	Ohio State University
Moore, James	Oregon State University
Payne, Victor W.E.	Payne Engineering
Reed, Sherwood C.	Environmental Engineering Consultants
Reddy, Ramesh	University of Florida
Schueler, Thomas R.	Center for Watershed Protection
Schwartz, Larry	Camp Dresser & McKee
Stiles, Eric	Bureau of Reclamation
Tchobanoglous, George	University of California, Davis

Due to the large number of contributors and the monumental efforts made by so many wetland scientists and practitioners in the development of this document, participation in the drafting and review process for this technology assessment cannot, and does not constitute approval or agreement by any participant with the content of this final report.

Data Quality and Validation

Data related to wetland design, operation, and performance exist, but are variable with respect to quality. Some design information is estimated from plans and specifications and has not been confirmed by as-built field measurements. Thus, wetland area estimates may be inaccurate because of difficult construction conditions, berm erosion during and following construction, or imprecise aerial photo interpretation. Similarly, water depths are rarely measured at more than a few points, and topography due to final grade variation or due to natural wetland conditions results in depth estimates that may be questionable.

Water flow rates can be measured with considerable accuracy, given state-of-the-art equipment and adequate calibration techniques. However, few facilities have a high level of instrument sophistication, and many only routinely estimate inflows or outflows, and not both. Internal flows are rarely measured. Considerable error has been observed in flow measurements from many treatment wetland facilities.

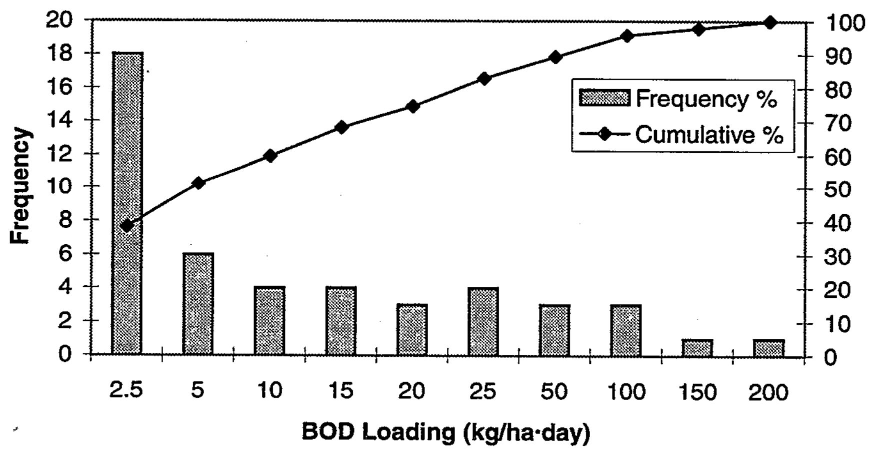
Numerous methods are available for analysis of water quality constituents. These methods tend to range from those requiring minimal sophistication to those methods employed in scientific research facilities. Significant variability exists in the accuracy of water quality data from different FWS treatment wetlands and analyzed at different facilities.

The inherent stochastic behavior of natural systems and the inescapable error introduced during the collection and analysis of wetland characterization and performance information is reflected in the NADB by data exhibiting a wide range of quality, accuracy, and precision. Some of the included data sets have large quantities of data collected from well-funded projects (i.e. large scale pilot projects), and the reported data is good quality and accurate. However, many data sets in the NADB have questionable flow rates and constituent concentration values, as is cautioned by the USEPA in the user instructions for the NADB. For this reason, greater or lesser reliance is warranted for conclusions formulated from different sites. When multiple data sets are included in an analysis, some of the uncertainty reflected in the results is likely due to measurement imprecision, while the rest is due to variables not included in the analysis.

The NADB was developed to identify sites and was not an attempt to analyze data or assess data quality. The NADB provides cautionary information on data quality with no attempt to review and reject questionable data. Although recognized errors have been corrected for this technology assessment, it is inevitable, due to the large amount of data presented that some of the results in this assessment will contain inaccuracies. The NADB operational data summary also disproportionately represents the southeastern US region and FWS treatment wetland systems with secondary or better quality influents. This situation needs to be considered carefully when attempting to draw conclusions regarding regional differences.

The majority of the systems in the database are lightly loaded systems with relatively low influent BOD and TSS concentrations. In several of these systems, the effluent BOD is greater than the influent BOD. Figure 2-1 shows that as of 1993, 50 percent of the systems (and over 70 percent of the observations) documented in the NADB had average organic loads of less than 5 kg BOD/ha·d. Approximately 28 percent of the systems measured had organic loads less than 1 kg BOD/ha·d. Only 21 percent of the systems documented received loading within the suggested range for secondary effluents from 12 to 50 kg/ha·d (calculated from hydraulic loading rate ranges suggested by Watson et al. in Hammer, 1989 for secondary and polishing treatment).

FIGURE 2-1
Influent BOD loading rates for FWS Wetland Systems in the NADB.



Most of the lightly loaded systems have effluent concentrations of BOD close to the influent concentration, and in some cases, the effluent BOD levels are higher than the influent. More than 44 percent of the influent BOD measurements for FWS wetlands in the NADB were less than 10 milligrams per liter (mg/L) (32 percent less than 5 mg/L). Nearly 60 percent of these systems had effluent BOD values less than 5 mg/L. Some of these systems with low effluent BOD were moderately loaded systems, but most were very lightly loaded.

Because of legitimate concerns about data variability, quality, or relevance, it is important to examine information from multiple systems; to look for consistent trends among systems and over time; and to question and understand conflicting results. It is also prudent to look to multiple, independent data sets to validate apparent trends and conclusions. The level of confidence in the conclusions stated in this report is proportional to the availability of corroborating evidence and is indicated, when appropriate, throughout the text.

SECTION 3

Wetland Processes

Free water surface (FWS) treatment wetlands are typically shallow vegetated basins. They are designed and constructed to exploit physical, chemical and biological processes naturally occurring in wetlands that provide for the reduction of organic material, total suspended solids, nutrients, and pathogenic organisms. FWS treatment wetlands take advantage of these natural treatment processes by providing time for settling and for the wastewater to contact the many different reactive surfaces found in wetlands. Wastewater normally has higher nutrient concentrations than natural wetland influents, thus, many of the wetland processes and constituent reductions proceed at increased rates. The increased nutrient loads delivered to treatment wetlands generally result in higher levels of biological production than that which occurs in natural wetlands receiving non-wastewater inputs.

Important wetland processes, as they relate to FWS constructed wetlands, are summarized in this section. Topics discussed include wetland hydrology, hydraulics, biogeochemistry, temperature effects, constituent characteristics, and aquatic vegetation. The intent of this section is to provide the reader with a brief introduction to wetland processes. For more detailed discussion of such processes, the reader may refer to the books available on wetlands (see Table 2-3).

Wetland Hydrology

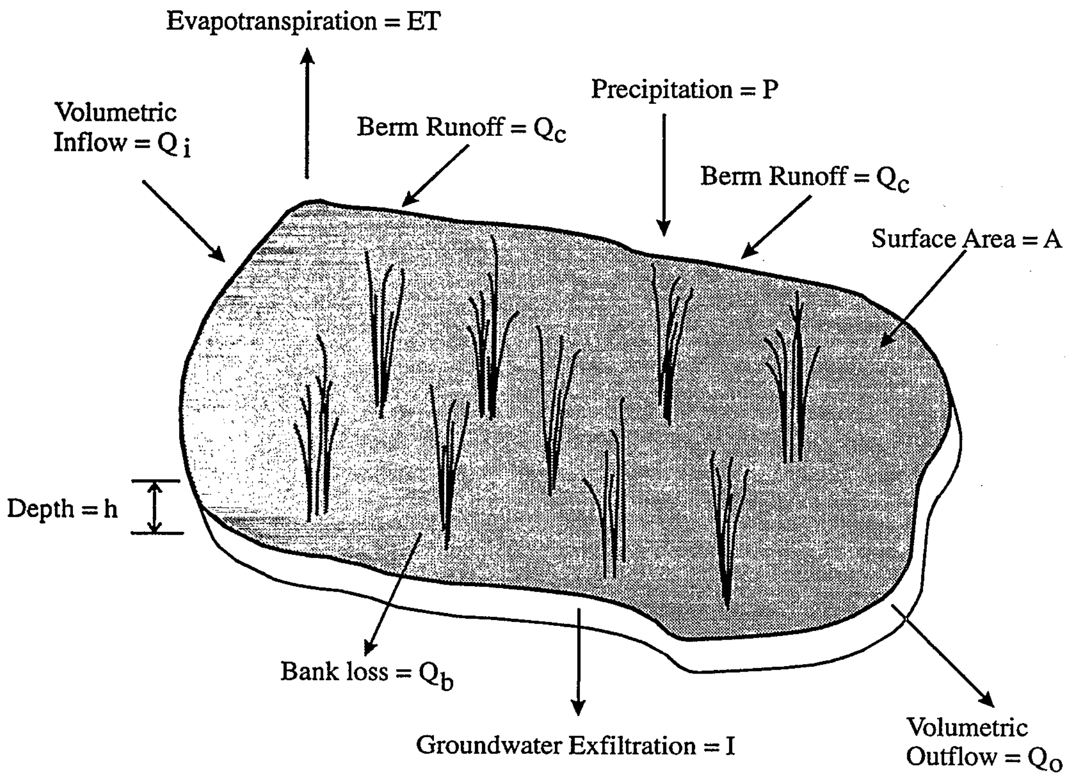
The hydrology of FWS wetlands, both natural and constructed, is often considered the most important factor in maintaining wetland structure and function, determining species composition, and developing a successful wetlands project (Mitsch and Gosselink, 1993; Hammer, 1992; Kusler and Brooks, 1988). Wetland hydrology directly influences and controls abiotic factors, such as water and nutrient availability, aerobic and anaerobic conditions in both the soil and water columns, water chemistry, soil salinity, soil conditions (e.g. peat building), and water depth and velocity. In turn, biotic components of a wetland (primarily vegetation) directly influence wetland hydrology through processes, such as transpiration, interception of precipitation, peat building, shading, wind blocks, and development of microclimates within the wetland. The development of a water balance or budget, the standard method for characterizing wetland hydrology, is described below.

Water Balance

The wetland water balance is used to quantify inflows and outflows of water to and from the wetland, and the wetland volume or storage capabilities. The flows and storage volume control the length of time water spends in the wetland, and thus the opportunity for interactions between waterborne substances and the wetland ecosystem. A thorough understanding of the dynamic nature of the wetland water balance, and how this balance affects pollutants, is necessary for the planning and design of FWS constructed wetlands.

Water enters natural wetlands via stream inflow, runoff, groundwater discharge and precipitation, and water is lost through stream outflow, groundwater recharge (infiltration), and evapotranspiration (Figure 3-1). These flows are extremely variable and stochastic in nature, which can cause large water level fluctuations to occur in natural wetlands.

FIGURE 3-1
Components of overall wetland water mass balance (Kadlec 1993).



In contrast, FWS constructed wetlands are typically isolated from stream inflows. Instead, their primary source of water is continuous wastewater inflow, precipitation and runoff, while water losses are via surface discharge through the outlet, evapotranspiration, and possibly percolation (if the wetland bottom and sides are unlined and/or permeable). The dominant steady wastewater inflow associated with FWS constructed treatment wetlands represents an important feature that distinguishes them from many natural wetlands. A dominant steady inflow, with little variation in water levels drives the ecosystem toward an ecological condition that is somewhat different from a stochastically driven system. Dry-out does not normally occur in FWS constructed wetlands, and only plants that can withstand continuous flooding will survive.

Although FWS constructed wetlands experience more constant inflows, seasonally variable wastewater flows can combine with seasonally variable precipitation and evapotranspiration to cause large differences in seasonal hydrologic functions. An overall water balance is required to perform the contaminant mass balance analyses necessary to predict or evaluate wetland functioning.

The averaging time period over which the water balance components are determined must be short enough (weekly to monthly) to capture seasonal effects. In addition, the averaging time period must be compatible with the frequency of water quality sampling. For instance, weekly water quality results would normally be combined with weekly average flows to determine mass removal rates. At a minimum, a detailed monthly or seasonal water balance, in which all potential water losses and gains are considered, should be conducted for any proposed FWS treatment wetland. An annual water budget will miss important seasonal wetland water gains or losses, such as heavy periods of winter precipitation or high summer evapotranspiration rates.

The overall dynamic water budget for a FWS constructed wetland can be stated as:

$$\frac{dV}{dt} = Q_i - Q_o + Q_c - Q_b + Q_{sm} + (P - ET - I)A \qquad (3\text{-}1)$$

where:

- dV/dt = rate of change in water volume (V) in the wetland with time (t), $[L^3/t]$
- Q_i = input wastewater flow rate, $[L^3/t]$
- Q_o = output wastewater flow rate, $[L^3/t]$
- Q_c = catchment runoff rate, $[L^3/t]$
- Q_b = bank loss rate, $[L^3/t]$
- Q_{sm} = snowmelt rate, $[L^3/t]$
- P = precipitation rate, [L/t]
- ET = evapotranspiration rate, [L/t]
- I = infiltration (or exfiltration) to groundwater, [L/t]
- A = wetland top surface area, $[L^2]$

Each term in this water budget may be important for a given constructed wetland, but rarely do all terms contribute significantly. The importance of the primary components of Equation 3-1 will need to be determined prior to the preparation of the wetland water budget. Some of the terms may be deemed insignificant and can be neglected from the water budget equation (e.g., Q_b, Q_c, Q_{sm}, are generally ignored). In addition, groundwater infiltration (I) can be neglected if the wetland is lined with some type of impermeable barrier.

Input Wastewater Flowrate

The daily influent wastewater flow (Q_i) will typically be the controlling inflow into a FWS treatment wetland. If wastewater flowrates are not known, they can be estimated using conventional engineering methods for predicting wastewater flows, such as water usage records, or user numbers and typical wastewater per capita flowrates found in the literature. The variability of wastewater flowrates may need to be considered when conducting wetland water balances, especially for small to medium size FWS treatment wetlands. Variable wastewater flows include seasonal peaks from vacation communities and high infiltration and inflow rates into collection systems, the latter being a condition that should be studied and minimized prior to treatment system design.

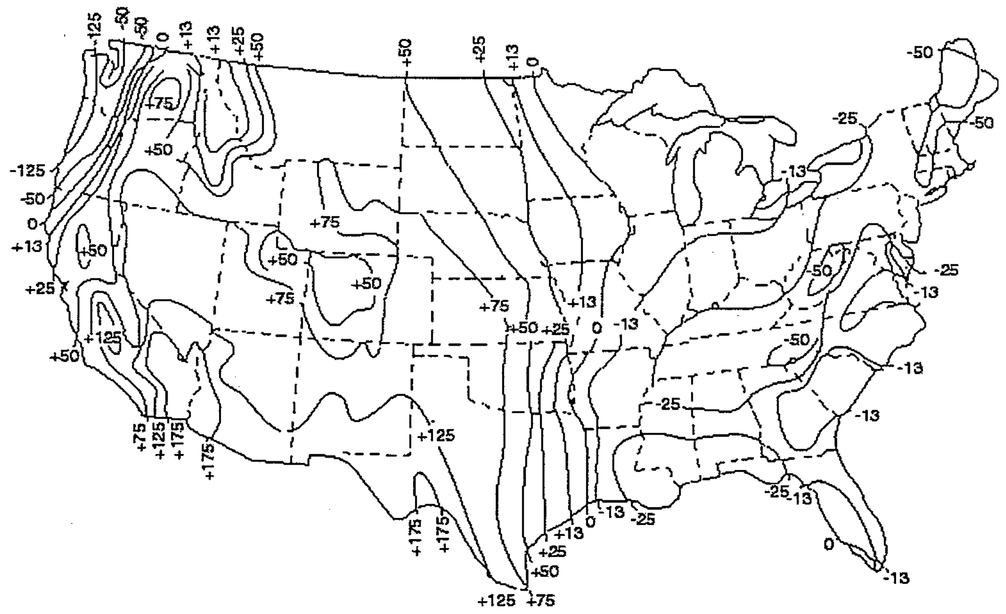
Precipitation

Depending on the time scale of the water budget, precipitation (P) data may be required in daily, weekly, monthly, seasonal, or annual quantities. Precipitation inflows into a wetland come from direct precipitation onto the wetland surface area, and runoff from the wetland catchment (i.e., berms and roads). The effects of precipitation on the wetland water balance can be significant, especially in areas of high rainfall or snowfall rates. High seasonal precipitation can dilute wetland pollutant concentrations, and the resulting effects may need to be considered in a wetland pollutant mass balance.

Evapotranspiration

Evapotranspiration in a FWS constructed wetland is the combined water loss due to evaporation from the water surface and transpiration from wetland vegetation. Many FWS wetlands operate with small hydraulic loading rates. For the 100 surface flow wetlands in North America, a hydraulic loading of 10.0 millimeters per day (mm/d) is found to be the 40th percentile (Knight et al. September 1993). Evapotranspiration (ET) losses approach a daily average of 5.0 mm/d in summer in the southern U.S.; consequently, more than half the water added daily may be lost to ET under these circumstances. Because ET follows a diurnal cycle, with a maximum during early afternoon and a minimum in the late nighttime hours, outflow from a FWS constructed wetland can cease during the day in areas of high ET rates. As shown in Figure 3-2, with the exception of the non-coastal, western U.S., annual water loss due to ET is largely replaced by precipitation.

FIGURE 3-2
Total annual losses (+) and gains (-) from evapotranspiration and precipitation in cm (ET-P) (Flach, 1973).



Output Wastewater Flow

The output wastewater flow (Q_o) corresponds to the amount of treated wastewater (effluent) leaving the FWS constructed wetland. The outlet in a FWS constructed wetland generally consists of some type of control structure that can be used to regulate water depth. Increasing or decreasing the water level also changes the wetland volume, which can influence the wetland water budget by providing more or less water storage potential.

Exfiltration to Groundwater (Infiltration)

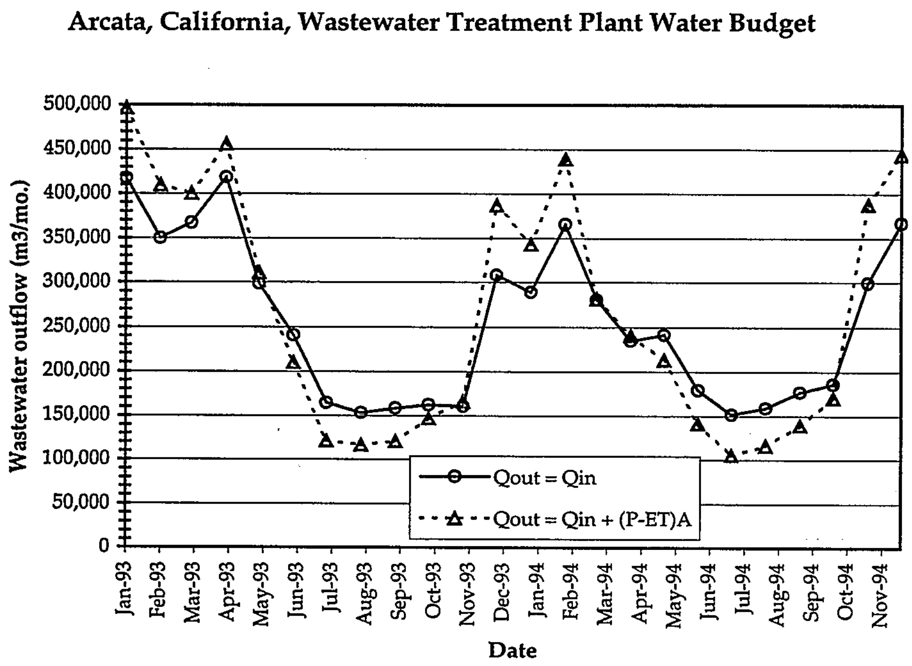
In a FWS constructed wetland, infiltration (I) is the loss of water that occurs into the bottom soils or berms. The effect is to reduce the amount of water remaining in a wetland and change the potential for each constituent transformation. Effluent constituent load(s) calculated at the surface discharge point from the wetland can be further reduced by the loss of certain soluble constituents as the water percolates from the system and infiltrates into the soil. If the FWS constructed wetland is lined with some type of impermeable barrier, percolation can be neglected in the water balance.

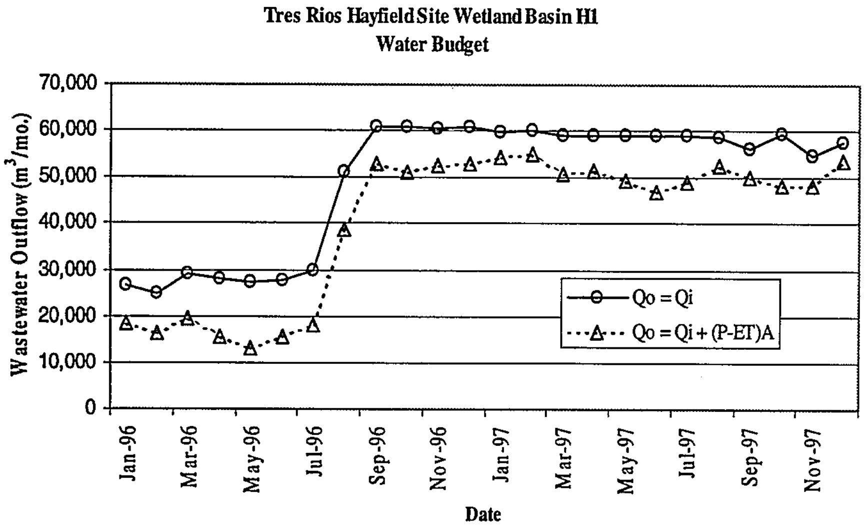
Meteorological Effects on Wetland Water Budget

FWS constructed wetlands generally have a more consistent hydrology than natural wetlands. However the variability of wastewater inflows, and the seasonal and stochastic nature of precipitation and ET can produce a variable seasonal hydrology in these wetlands.

The effects of precipitation and ET on monthly outflows from the Arcata, California, FWS constructed wetland system and the Phoenix, Arizona, Tres Rios Hayfield FWS treatment wetland H1 are shown in Figure 3-3. In reference to Arcata, California, the solid line is the wastewater outflow neglecting the effects

FIGURE 3-3
Monthly water budget for Arcata's wastewater treatment plant (Arcata, California) showing the effects of precipitation and evapotranspiration on the water budget in a Coastal wetland system and the monthly water budget for the Tres Rios Hayfield Site basin H1 (Phoenix, Arizona) showing the influence increased ET and reduced precipitation has in arid regions.





of precipitation and ET ($Q_o = Q_i$). Note the increase and variable nature of wastewater inflows for the months of December through April, which is caused by seasonal infiltration and inflow into the collection system. The dashed line indicates the wastewater outflow when monthly precipitation and ET are included in the water budget ($Q_o = Q_i + (P-ET)A$). Precipitation increases wastewater outflows during the wet weather season (November through April) whereas ET reduces wastewater outflows during the warmer months of the year (May through October). In contrast to coastal and the more temperate regions of the country, Phoenix, Arizona, has an annual precipitation of about < 25 cm/yr while ET can be as high as 1.2 cm/day in midsummer (AZMET, 1998). Water budgets both neglecting and considering these meteorological effects for a wetland in this area are shown below. The lack of precipitation allows the effects of ET to be seen particularly in the May through July time periods. Consideration of meteorological effects on the water budget is warranted in this case, as the reduction in surface outflow from these systems as a result of ET can result in the degradation of quality due to the evaporative concentration of salts.

Wetland Hydraulics

Wetland hydraulics is the term applied to the movement of water through the wetland. Improper hydraulic design can cause problems with water conveyance, water quality, odors, and vectors. For example, in a few instances FWS constructed wetland design has failed to account properly for head loss, with inlet over-flooding as the result. Important wetland hydraulic definitions and basic wetland hydraulic principles are presented and discussed below.

Wetland Hydraulic Definitions

Water Depth

Compared to other aquatic treatment systems (lagoons for example), a wetland can be designed to incorporate features that allow the system to be operated over a wide range of depths from less than 10.0 cm to 1.5 m (4 in. to 5 ft). Depending on bottom topography and slope of the water surface, the water depth will not be equal at all locations in a constructed wetland. For natural wetlands and some large FWS constructed wetlands, accurate determinations of water depth may be difficult due to lack of survey data. However, many FWS constructed wetlands are designed and constructed with strict engineering grade control and detailed surveys. Consequently, the elevations of the bottom, berms, islands, and inlet and outlet structures are known with some degree of accuracy. With detailed elevations, accurate estimates of average water depth can be obtained. Water depth in FWS constructed wetlands should be considered an operational characteristic as well as a design characteristic. The effective depth of a wetland will change with time as litter fall below the water surface and detritus buildup on the bottom begin to reduce the depth, therefore, reducing the effective hydraulic volume.

Surface Area

The term surface area (A) can embody at least two different concepts when considering FWS constructed wetlands. First, surface area can refer to the wetted projection of the constructed wetland in plan view. This is relatively easy to define using construction or "As-Built" drawings. If this information is not available, aerial photography or a survey of the wetland water surface perimeter can be conducted and produce accurate estimates. For most situations, the surface area or the wetland footprint at the water surface is a good estimate of the wetland bottom area. In the standard use of the term surface area, either a contaminant mass or a depth is used to define effectiveness and utilization of the system resulting in terms such as kg/ha/day (lbs/acre/day) or cm/ha/day (ft/acre/day).

Second, surface area can be referred to as the effective surface area, or the amount of area that comes in routine contact with the water. The effective area is available for the sorption of pollutants, or the attachment of microbial communities. Although it would be appropriate for use in modeling the performance of such systems, the effective surface area of a FWS wetland is difficult to quantify. Not only does it include the hydraulically active portion of the wetland bottom, but also the submerged surfaces of vegetation, litter, and detritus.

Volume

The volume (V) of a FWS treatment wetland is the potential quantity of water (neglecting vegetation, litter and peat) found in the wetland basin. As indicated under the depth discussion, the volume changes with time for a given outlet weir setting.

Wetland Porosity or Void Fraction

In a natural or constructed wetland, the vegetation, litter, and detritus occupy a portion of the water column, thereby reducing the space available for water. The porosity of the wetland (ε), or void fraction, is the ratio of the theoretical or empty basin volume to the actual volume available for water to occupy in a wetland. Wetland porosity can be difficult to determine as it varies in the x-y (horizontal) dimension due to plant species composition and distribution, and in the vertical direction with lesser values near the bottom in the litter layer. As a result, wetland porosity values listed in the literature can be highly variable and, sometimes, not in good agreement. In a recent study, emergent vegetation was found to occupy between 3 percent and 8 percent of the available volume depending upon species and stem density (Lagrace et al., 2000). Literature values as reported in Reed et al. (1995), shows wetland porosity values ranging from 0.65 to 0.75 for vegetated wetlands, with lower numbers for dense mature wetlands. Finally, Kadlec and Knight (1996) report that average wetland porosity values are usually greater than 0.95, and as such, $\varepsilon = 1.0$ can be used as a good approximation.

The overall effect of porosity is to reduce the wetland volume available for water flow and storage. In turn, this reduction in volume reduces the amount of time water remains in the wetland, and the potential for constituent removal to occur. Lower wetland porosity values correspond to a lower fraction of the wetland volume available for water, shorter hydraulic detention times, lower removal efficiencies, and result in larger required wetland areas to achieve desired treatment goals. To be conservative, a porosity (ε) value of 0.7 to 0.9 could be used in FWS constructed wetland design calculations, with lower ε values for densely vegetated wetlands, and higher ε values for wetlands with more open water areas.

Volume is not the only factor affected by vegetation density and porosity, head-loss is equally important. The friction coefficient that controls head-loss through the wetland depends on the vegetation density. Highly vegetated areas will have a greater head-loss than open areas, and this increase may cause a significant backwater effect and can lead to the development of preferential flow paths. If this potential backwater is not accounted for in the FWS wetland design, inlet flooding may occur as the wetland vegetation matures, density increases, and the porosity decreases.

Hydraulic Detention Time

The theoretical (or nominal) hydraulic detention time (t) is the ratio between flowrate and the wetland volume available for water flow, and includes the volume reducing effects of vegetation (porosity). The theoretical hydraulic detention time can be calculated as:

$$t = \frac{V\varepsilon}{Q} \qquad (3\text{-}2)$$

where:

	t	=	hydraulic detention time, [t]
	V	=	volume of wetland basin, [L^3]
	ε	=	wetland porosity, and
	Q	=	flowrate, [L^3/t]

The flowrate (Q) value used in the hydraulic detention time calculation is generally one of two values: input wastewater flowrate (Q_i) or average flowrate (Q_{avg}). The use of input wastewater flowrate (Q_i) in Equation 3-2, results in the inlet hydraulic detention time. The inlet hydraulic detention time neglects the effects of precipitation, evapotranspiration, and infiltration, and assumes $Q_i = Q_o$. The input wastewater flowrate (Q_i) should only be used for preliminary calculations, or when no measurement or estimate (i.e. water balance) of the output wastewater flowrate (Q_o) exists.

A more realistic measure of detention time can be computed using the average flowrate (Q_{avg}) in Equation 3-2 to account for the effects of water gains and losses (precipitation, evapotranspiration and infiltration) that occur in a wetland. The average flowrate can be estimated by:

$$Q_{avg} = \frac{Q_i + Q_o}{2} \tag{3-3}$$

Accuracy of the theoretical hydraulic detention time calculation is dependent on the measurements of depth, surface area, and the estimate of porosity. As mentioned earlier, the theoretical detention time may also be a very poor estimate of the actual hydraulic detention time due to hydraulic short-circuiting (e.g., preferential flow paths). The modal detention time, which can be determined by a tracer study, will always be shorter than the theoretical value (sometimes less than half).

Hydraulic Loading Rate

The hydraulic loading rate (q) is the rainfall equivalent of whatever flowrate is under consideration; however, it does not imply the physical distribution of water uniformly over the wetland surface. The hydraulic loading rate is defined as:

$$q = \frac{Q}{A} \tag{3-4}$$

where:

where:	q	=	inlet hydraulic loading rate, [L/t]
	Q	=	flowrate, [L^3/t]
	A	=	wetland surface area, [L^2]

When the input wastewater flowrate (Qi) is used in Equation 3-4, the resulting calculation is for the inlet hydraulic loading rate, which neglects the effects of other hydrologic inputs and outputs such as precipitation, infiltration, and evapotranspiration. Like hydraulic detention time, the average flowrate (Q_{avg}) can also be used in Equation 3-4, resulting in the average hydraulic loading rate, accounting for water losses and gains in the wetland.

Water Conveyance

Water conveyance in FWS wetlands is complex hydraulically, varying in both space and time due to changing inflow conditions and the stochastic nature of hydrologic events. Water moves through FWS wetlands in response to a surface water gradient from inlet to outlet, impeded by the friction created from submerged plants, litter, peat, and the bottom and sides of the wetland. Some type of outflow structure, such as an adjustable weir, typically is used to control the water depth. The hydraulic profile of the water surface is dictated by these factors, combined with the bottom slope and length-to-width ratio of the wetland.

It is important to consider wetland hydraulics when designing a FWS constructed wetland. The primary concern is to ensure that the wetland can handle all potential flows without creating significant backwater problems, such as flooding the inlet structures or overtopping of berms.

Assessment of the head-loss from inlet to outlet can usually be done using Manning's equation. When a more detailed head-loss calculation is required, or the effects of precipitation, evapotranspiration, and infiltration need to be considered in water conveyance calculations, then the simplified one-dimensional flow procedure presented by Kadlec and Knight (1996) can be used. In the case of complex geometry or irregular boundaries, more detailed hydraulic modeling approaches may be required, such as the one-dimensional HEC2 or HECRAS (U.S. Army Corps of Engineers Hydraulic Engineering Center), or the two-dimensional Surface Water Modeling System (Engineering Computer Graphics Laboratory, Brigham Young University).

Aspect Ratio

The aspect ratio is defined as the quotient of the average length of the major axis and the average width of a wetland. Because the footprint of a wetland can have a variety of shapes, it is the effective aspect ratio between inlets and outlets that is important. In general, FWS treatment wetlands with high length to width ratios are of greatest concern with respect to head-loss. However, some early researchers reported that the treatment performance of FWS constructed wetlands is better at higher aspect ratios (Wile et al. 1985).

For wetland systems with high length to width ratios, careful consideration needs to be given to increasing costs associated with more lineal feet of berm construction, head-loss, and the internal flow through the wetland. In some instances weir overflow rate, location of inlets and outlets, and elevation of berms may be as important or more important than the influence of the aspect ratio on wetland performance.

Internal Flow Patterns Effects/Physical Facilities

The low gradients found in FWS treatment wetlands result in very low water velocities, approaching laminar flow in highly vegetated areas. This type of flow regime produces quiescent conditions, an ideal situation for many of the physical, chemical, and biological processes that occur in FWS wetlands.

Water does not flow through a FWS wetland in one flow direction or path. Instead, water flows through a complex maze of submerged vegetation, litter, detritus, and other obstructions (e.g., islands); forcing the water velocity to increase and decrease and to continually change direction. Water in open areas located away from submerged vegetation or accumulated bottom material is less subject to friction and generally moves at faster velocities than water located in densely vegetated areas. Open water zones are subject to wind-driven surface flows, which can move at higher velocities than water below the surface, and cause mixing to occur at different depths. Some areas of a FWS constructed wetland, such as corners and behind islands, may become isolated from the main flow path, creating pockets of dead space for which no or little water exchange occurs. The bottom topography may also form deeper pockets or pools, creating more dead space zones, resulting in a constantly changing internal flow pattern intermediate between the ideal extremes of plug flow and complete mixing.

All of these processes combined cause water to flow through a FWS wetland in a shorter time period than defined by the theoretical hydraulic detention time. In many cases, water can flow at high velocities through a small portion of the total wetland volume, significantly lowering the hydraulic detention time, a process often referred to as short-circuiting. For example, the theoretical detention time for the Boggy Gut treatment wetland was estimated to be 19 days; however, the measured value using tracer studies was approximately 2 days (Knight and Ferda 1989). Careful consideration of the site characteristics showed that this difference was due to large zones (volumes) of wetland (dead zones) that were not incorporated effectively in the treatment of the influent flow.

The placement, size, and orientation of inlet and outlet works can be an important factor in determining the hydraulic response of a FWS constructed wetland to wastewater inputs and process withdrawals. Experience to date has been to distribute the influent over a large portion of the inlet region and to place relatively narrow (0.6 to 1 m) rectangular adjustable weirs along the discharge region of the cells. To mitigate some of the short-circuiting inherent in FWS wetlands, several strategies exist for providing a collection volume at the terminus region of the wetland (Kadlec and Knight 1996). In one approach, a deeper zone is created in the outlet area with the outlet weir control structure placed away from the bank into the collection volume. Other approaches have been to collect the influent in vegetated shallow water zones outfitted with barriers (fenced) to minimize fish and amphibian export with the effluent. Recently, square non-adjustable weir structures have been experimented with to increase weir overflow rates from 225 to 500 liters per meter of weir length per minute (L/m·min) over that of more conventional outlet weir designs (Gearheart 1998, Unpublished data).

Water Balance Effects on Wetland Hydraulics and Water Quality

The variability inherent in wastewater flowrates and the stochastic nature of meteorological events controls wetland hydraulics, which in turn affects treatment wetland performance and water quality. Impacts to wetland hydraulics can best be described by noting the increases and decreases to the wetland hydraulic detention time caused by water gains and losses in the wetlands water balance. Likewise, the wetland hydraulic detention time can be used to explain water balance impacts to wetland water quality.

Precipitation to a wetland increases inflow, which affects wetland hydraulics by decreasing the hydraulic detention time, and affects water quality by diluting constituent concentrations. The combination of these two influences can provide either poorer or better performance of the wetland with regard to water quality. In systems receiving low influent constituent concentrations, concentration reduction is likely to be less evident with precipitation additions; in heavily loaded systems, concentration reductions will often be more notable. In both cases, mass load reduction could be poorer with precipitation additions because the added flow reduces the effective hydraulic detention time.

Evapotranspiration has the effect of increasing hydraulic detention time and increasing constituent concentrations. The combination of precipitation and evapotranspiration can improve concentration reduction in very lightly loaded systems, but generally decreases concentration reduction in heavily loaded systems. The effect of exfiltration is similar to evapotranspiration by increasing the hydraulic detention time and increasing the potential for constituent removal. Constituent load reduction can further be enhanced by the loss of constituents with the water as it infiltrates into the soil.

Wetland Biogeochemistry

Free water surface treatment wetlands support a variety of sequential and often complementary treatment processes. The predominant physical, chemical, and biological mechanisms operative in FWS treatment wetlands are summarized in Table 3-1. These interrelated biological, chemical, and physical treatment processes control the transport and transformation of constituents through FWS wetlands. Specific processes controlling total suspended solids, biological oxygen demand, chemical oxygen demand, dissolved oxygen, nitrogen, dissolved organic phosphorus, pH, organic pollutants, and metals within FWS wetlands are more explicitly described in subsections below.

A hypothetical partitioning of treatment processes throughout the wetland volume is shown in Figure 3-4. Wetland treatment processes are generally associated with vertically and horizontally differentiated zones within the wetland volume. These zones are linked both hydro-dynamically and through sequential physical, chemical and biological reactions. In the inlet zone, the physical process of sedimentation dominates treatment and quickly removes the easily settleable solids and their associated constituent. Finer particulates are removed slowly by flocculent settling farther into the wetland.

The location of various aerobic biological processes operating within the wetland is partially determined by the dissolved oxygen concentration. The oxygen demand from degradable carbon compounds is met near the surface of the open water zones of the wetland where oxygen transfer from the atmosphere and release to the water column by photosynthesis is greatest. Oxidation of ammonia nitrogen (nitrification) in a wetland occurs where carbonaceous BOD has been generally satisfied and sufficient dissolved oxygen is present in the water column.

Denitrification, or reduction of the nitrate nitrogen species, has been shown to be a significant process in FWS constructed wetlands. The combination of anoxic conditions, physical substrates for microbial attachment, and internal carbon sources provide ideal conditions for nitrate conversion to dinitrogen gas. The dissolved organic carbon produced as a by-product of detrital decomposition supplies the carbon for this microbial process. Because most denitrifying

TABLE 3-1
Mechanisms and factors that affect the potential for removal or addition of water quality constituents in FWS wetlands (Adapted from Stowell et al. 1980).

Mechanism	Water Quality Constituent*							Description
	BOD	TSS	N	P	DO	Bacteria Virus	Heavy Metals	
Physical								
Absorption			S		P/S			Gas transfer to and from water surface
Adsorption/ desorption	I	S				P	I	Interparticle attractive force (van de Waals force); hydrophilic interaction
Emulsification		S					S	Suspension of low solubility chemicals
Evaporation						I	S	Volatilization and aerosol formation; thermal moderation
Filtration Impaction	I	S					I	Particulates filtered mechanically as water passes through substrate and plants
Flocculation	P	P				P	S	Interparticle attractive force (van de Waals force); hydrophilic interaction
Photochemical reactions								Solar radiation is known to trigger a number of chemical reactions. Radiation in the near-ultraviolet (UV) and visible range is known to cause the breakdown of a variety of organic compounds. Pathogenic bacteria and virus attenuation.
Sedimentation	P	P	I	I	I	S	P	Gravitational settling of larger particles and contaminants
Thermal	I		P	S				Autoflocculation; natural coagulants
Volatilization			P					Similar process to gas absorption, except that the net flux is out of the water surface.
Chemical								
Adsorption				P		S	S	On substrate and plate surfaces
Chelation				S			P	Formation of complex metal compounds through ligands
Chemical reactions								Hydrolysis, for example, is an important chemical reaction that occurs in the environment, by which proteins are converted into amino acids and other soluble compounds. Organic nitrogen can also be converted to ammonium.
Decomposition						P		Decomposition or alteration of less stable compounds by phenomena such as UV irradiation, oxidation, hydrolysis
Oxidation/ reduction reactions	P	S					P	Anoxic condition; metal speciation; organic acid production
Precipitation				P			P	Formation of co-precipitates with insoluble compounds

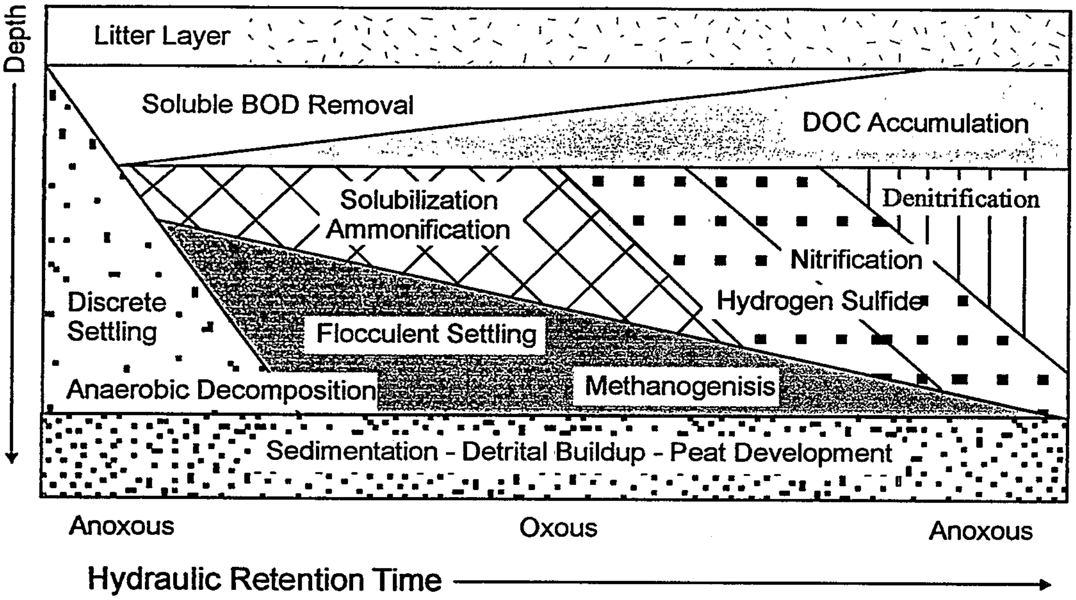
Mechanism	BOD	TSS	N	P	DO	Bacteria Virus	Heavy Metals	Description
	Water Quality Constituent*							
Biological								
Algal synthesis			S	S				The synthesis of algal cell tissue using the nutrients in wastewater.
Assimilation, plant	C	C	S	P/S	I/C	I	S	Uptake and metabolism by plants; root excretions may be toxic to enteric organisms; transpiration concentrates effluent; dissolved oxygen supply
Bacteria/ Metabolism								Removal of colloidal solids and soluble organics by suspended, benthic and plant supported bacteria; bacterial nitrification, denitrification; microbial mediated oxidation
Aerobic	P/C	S	I	I	P	P		
Anaerobic		P/C	C	C				
Plant adsorption			S	S	C		S	Under proper conditions, significant quantities of contaminants will be taken up by plants.
Predation		P				S		Zooplankton and aquatic insect larva particles; odonata and fish-aquatic insect

Notes: *P = primary processes, S = secondary processes, I = incidental effect (occurring with removal of other constituent), C = contributory effect, S/P = depends on influent and design conditions, N = negative.

bacteria are obligate anaerobes, oxygen must be suppressed to less than 0.5 mg/L in the water column. Both nitrification and denitrification processes are temperature dependent, and enzymatically shut down at temperatures less than 5-7 °C.

This brief introduction illustrates how FWS constructed wetlands incorporate a similar sequence of physical, chemical, and biological treatment processes to those commonly employed in conventional wastewater treatment. FWS constructed wetlands can be designed to emphasize some treatment processes over others by altering the geometry, hydraulics, and plant types, densities, or locations. A more detailed discussion of the role of unique features of FWS constructed wetlands and the processes controlling specific constituents of interest follows.

FIGURE 3-4
Conceptual partitioning of treatment processes through a FWS wetland (Gearheart, 1998).



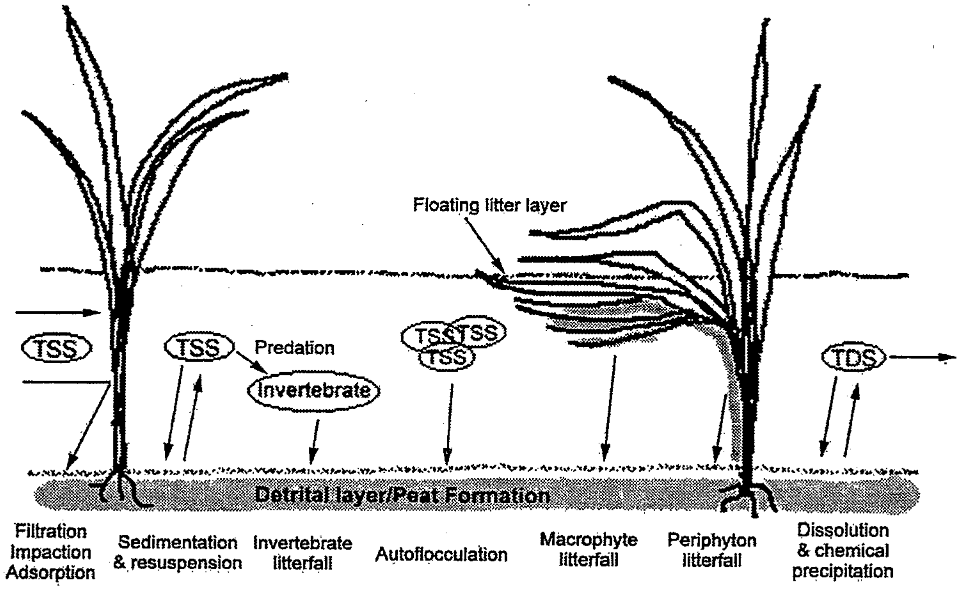
Total Suspended Solids

Processes

Total suspended solids (TSS) are both removed and produced by natural wetland processes. During treatment, settleable incoming particulate matter usually has ample time to settle and become trapped in litter or dead zones. Once there, soluble organic constituents are reduced to carbon dioxide and low molecular weight organic acids and inorganic constituents can become bound as sulfide complexes or become buried through sediment accretion. Wetland scientists generally refer to the combination of removal processes as filtration, although stem and litter densities are not typically high enough to act as a filter mat. As shown in Figure 3-5, a number of wetland processes produce particulate matter including: death of invertebrates, fragmentation of detritus from plants and algae, and the formation of chemical precipitates such as iron sulfide. Bacteria and fungi can colonize these materials and add to their mass.

In wetlands, velocity-induced re-suspension is minimal, but gas lift and bioperturbation can reintroduce solids into the water column. Wetland sediments and micro-detritus are typically near neutral buoyancy, flocculent, and easily disturbed. Bioperturbation by fish, mammals, and birds can re-suspend these materials and lead to additional TSS in the wetland effluent. The oxygen generated by algal photosynthesis or methane formed in anaerobic processes can cause flotation of floc assemblages. Re-suspension due to fluid shear forces on bed solids is not usually a significant process except in the vicinity of a point discharge into the treatment wetland.

FIGURE 3-5
Wetland TSS removal, re-suspension, and internal generation processes.



The magnitude of wetland particulate cycling is large, with high internal levels of gross sedimentation and re-suspension, almost always overshadowing TSS influent loading. TSS effluent concentrations rarely result from an irreducible fraction of the influent TSS, and are often dictated by the wetland processes that generate TSS within the wetland. Most FWS constructed wetland designs are determined by an effluent limitation other than TSS and are sufficiently large that effluent TSS approaches background levels. Large expanses of open water not followed by vegetation can however lead to excessive algal growth and subsequent high effluent TSS.

High incoming TSS or high nutrient loads that stimulate high TSS production may eventually lead to a measurable increase in bottom elevation (van Oostrom and Cooper 1990). In lightly loaded FWS wetlands typical accretion rates ranged from 2 to 10 mm/yr (Richardson et al, 1994). Solids accretion rates were a function of distance from the inlet and vegetation density for measurements obtained in 12 experimental marshes receiving oxidation pond effluent in Arcata, California, from September 1979 through September 1982. In year 3 of operation, the solids bank had extended approximately 10 to 15 meters (12.5 – 20 percent) into the cells from the influent point. When measured next to clumps of vegetation, the depth of settled solids varied from 20 to 36 cm, while in open areas, this was reduced to 4 to 10 cm after the 3 years of operation (Gearheart et al., 1983). As yet, no treatment wetland has required maintenance because of normal solids accumulation, including some that have been in operation for 20 years or more, but this is unlikely to last indefinitely. In situations of high incoming non-volatile solids, a settling basin can be designed to intercept a large portion of the solids, thus providing for easier cleanout and extending the life of the inlet region of the wetland.

Settleable Solids Reduction-Anaerobic Decomposition

The benthic decomposition of accumulated solids from the influent and from the plant litter produced in the wetland has a delayed effect on the oxygen budget and biochemical oxygen demand (BOD) concentrations in a FWS treatment wetland. The accumulated material compacts and increases in density as anaerobic processes release aerobically degradable by-products to the sediment and organic layer pore water. These aerobically degradable by-products subsequently diffuse into the overlying water column and add to the BOD.

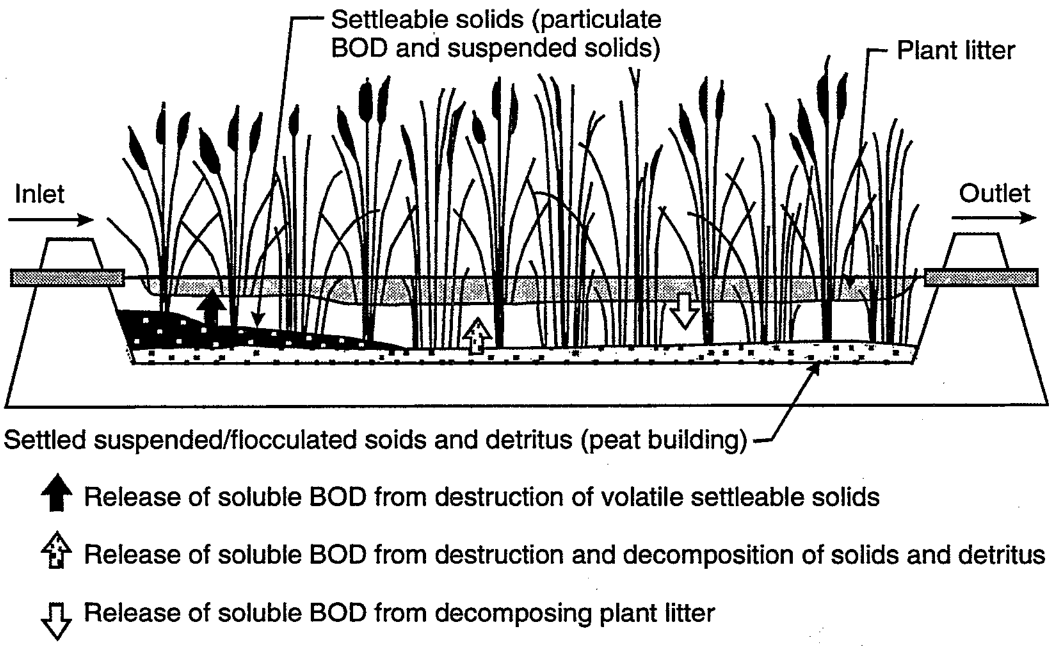
Accumulated organic debris degrades at different rates depending on the source and composition of the material. As the degradability of the material decreases, the decomposition rates slow and the nature of the soluble by-products change. The implication of this degradation rate and its relationship with the BOD in the water column is significant. The half-life of soluble BOD is approximately 3 days while the half-life of organic sediment, which is temperature-dependent, is closer to 4 months. Earlier observations by sanitary engineers of the role that benthic organic deposits played in the oxygen budget in streams is analogous in many ways to conditions in a FWS wetland. In streams, the oxygen requirements of benthic organic deposits are limited by the rate at which production of diffusible degradable material enters the overlying water column and not by the rate at which anaerobic breakdown occurs (Phelps 1944). For a given solids accumulation rate and temperature regime, a steady-state condition of organic sediment decay and release of soluble BOD to the water column should develop. This release of soluble BOD from the litter layer is one component contributing to the background BOD.

Biochemical Oxygen Demand

For FWS treatment wetlands receiving municipal wastewater, some fraction of the influent carbon compounds are dissolved while the rest enters in the form of particulate matter. Particulate settling provides one removal mechanism, and typically occurs in the inlet region of the wetland (Figure 3-6). Microbial communities process the dissolved carbon compounds. Microbial removal processes include oxidation in the aerobic regions of the wetland and methanogenesis in the anaerobic regions. Active microorganisms are usually associated with solid surfaces, such as litter, sediments, and submerged plant parts.

In addition to microbial decomposition, dissolved carbon is fixed into new biomass during photosynthesis. The decomposition of this biomass, litter and sediments produces a return flux of BOD to the water column. The balance between removal of influent BOD and the decomposition processes contributing BOD determines the wetland effluent concentration of this constituent.

FIGURE 3-6
Simplified portrayal of wetland carbon processing. Incoming BOD_5 is reduced by deposition of particulate forms and by microbial processing in floating, epiphytic, and benthic litter layers. Decomposition processes create a return flux.



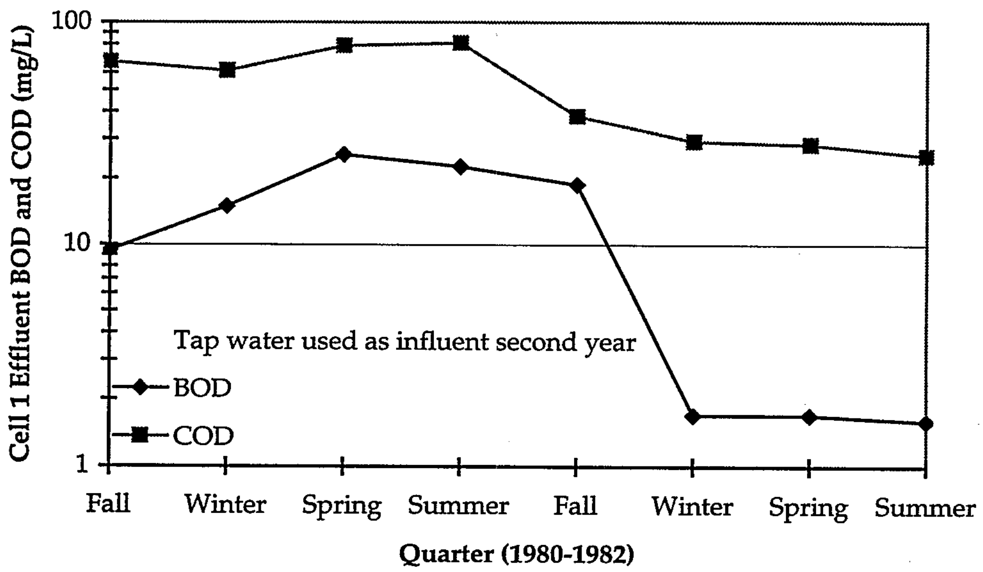
Chemical Oxygen Demand

The chemical oxygen demand (COD) measures the concentration of oxidizable compounds using a strong chemical oxidant. Thus, the COD test measures the sum concentration of two distinct fractions of oxidizable compounds: easily biodegradable compounds and oxidizable but not easily biodegradable compounds. The concentration of easily biodegradable compounds is often represented as BOD_5, the biological oxygen demand measured after 5 days of incubation, with the difference between the COD and BOD_5 representing the concentration of compounds that is not easily biodegradable. Some of these non-BOD compounds are degradable under anoxic conditions via anaerobic decomposition, or under aerobic conditions in periods of longer than 5 days.

Physical and microbial processes remove COD while other processes produce COD in FWS constructed wetlands. Effluent COD concentrations from the Arcata wetland cells only varied from 60 to 66 mg/L while the influent BOD ranged from 45 to 92 mg/L (Gearheart et al., 1983). Consistent COD effluent concentrations from the pilot wetland cells, even with a ten-fold range in hydraulic/organic loading, indicate that the effluent concentrations are more closely associated with the amount and type of aquatic plants decaying within the wetland than the influent BOD load. The COD/BOD ratio averaged 3.7 for the influent (oxidation pond effluent) while the wetland effluent COD/BOD ratio varied from 3.1 at the beginning of the study to 28 at the end of the study.

In another study, a pilot cell was loaded at 50 kg/ha·d for 15 months, after which time the influent was switched to tap water for 9 months. The concentration of BOD and COD before and after the addition of tap water (no addition of influent BOD) is shown in Figure 3-7. The COD/BOD ratio was 3.9 during the BOD loading period.

FIGURE 3-7
BOD and COD effluent concentration before and during tap water loading to Arcata Pilot Project wetland.



After the switch to freshwater, the COD/BOD ratio increased to 17 with COD and BOD concentrations of 30 and 1.7 mg/L, respectively. It appears, based on these observations, that the detrital material contributed about 1.7 mg/L of BOD at the wetland cell hydraulic loading rate of 240 mm/day.

Dissolved Oxygen

Dissolved oxygen is depleted to meet wetland oxygen requirements in four major categories: sediment/litter oxygen demand, respiration requirements, dissolved carbonaceous BOD, and dissolved nitrogenous oxygen demand NOD. Sediment oxygen demand is the result of decomposing detritus generated by carbon fixation in the wetland, and the decomposition of precipitated organic solids that entered with the wastewater. NOD is exerted primarily by ammonium nitrogen, but ammonium may also be contributed by the mineralization of organic nitrogen. Decomposition processes in the wetland also contribute to NOD and BOD, further increasing the oxygen demand and reducing the dissolved oxygen in the wetland water.

Plant roots also require oxygen, which is normally transported downward through passages (aerenchyma) in stems and roots. Some surplus of oxygen may be released from small roots into their immediate environs, but it is quickly consumed by the local oxygen demand (Brix 1994a). Wetland soils are typically anoxic or anaerobic (Reddy and D'Angelo 1994).

Wetland open-water areas can be aerated via oxygen transfer from the atmosphere at the air-water interface. Reaeration mechanisms include dissolution and diffusion (O'Connor and Dobbins 1958), as well as turbulent transfer associated with rainfall and wind induced surface mixing (South Florida Water Management District [SFWMD] unpublished data). In un-shaded open water areas, photosynthesis by algae within the water column produces oxygen, sometimes creating dissolved oxygen concentrations in excess of the saturation limit (Schwegler 1978). Photosynthesis stops at night, and respiration, which consumes oxygen, then dominates. The result is strong diurnal variations in water column DO for lightly loaded, algae-rich, open water wetlands.

In vegetated regions of the wetland, shading prevents high algal concentrations and DO levels are typically low near the surface. Anoxic or anaerobic conditions persist throughout the remainder of the water column. The effect of vegetation on DO level in the Arcata Enhancement Marsh is shown with the DO in the non-vegetated zones (Figure 3-8) significantly higher than that in the vegetated zone (Figure 3-9).

FIGURE 3-8
Vertical distribution of DO in a submergent plant zone (depth = 1.0 m) of the Arcata Enhancement Marsh.

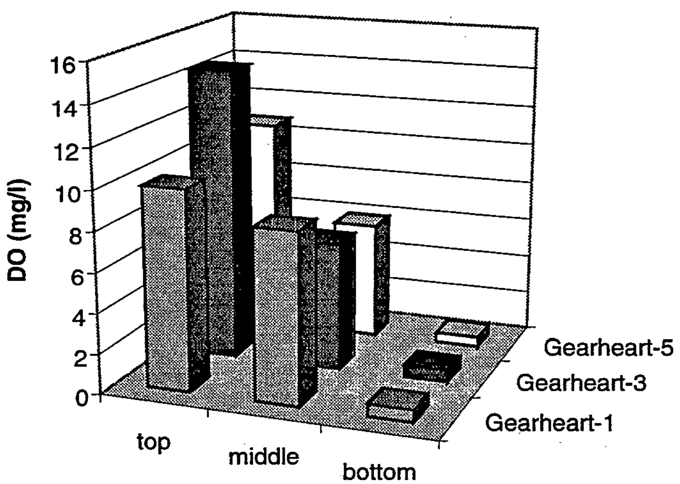
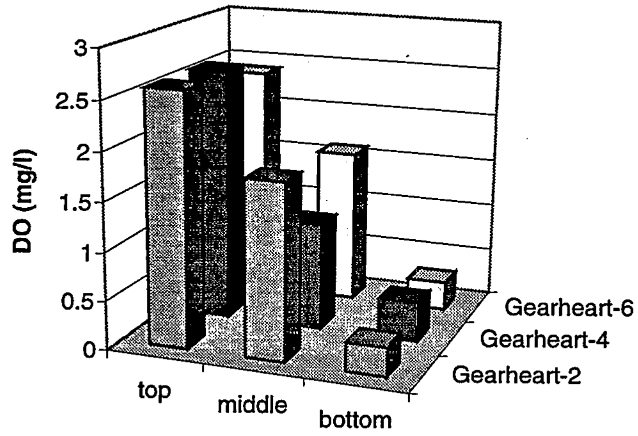


FIGURE 3-9
Vertical distribution of DO in an emergent plant zone (depth = 1.0 m) of the Arcata Enhancement Marsh.



Nitrogen

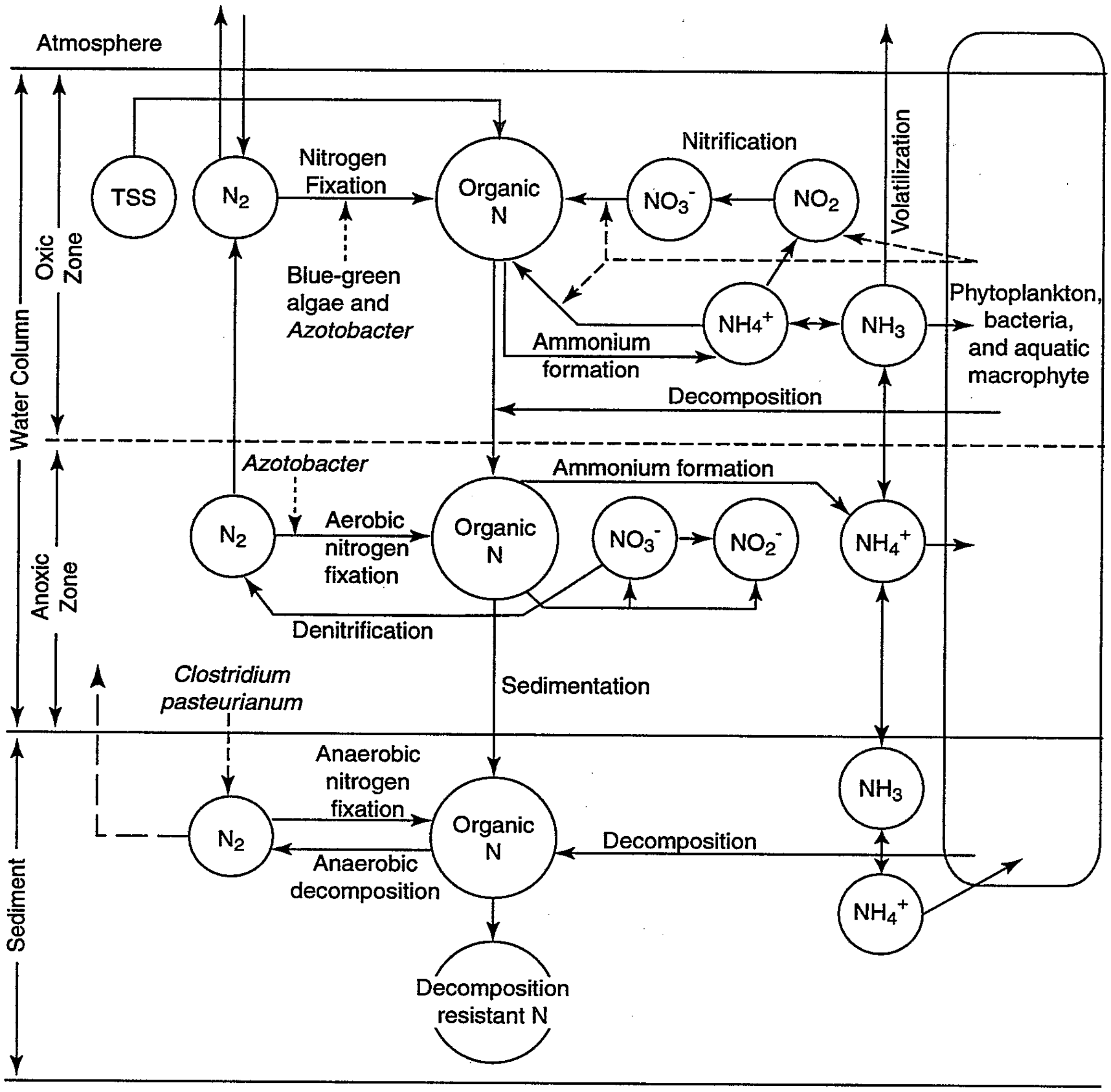
Nitrogen is a key element in biogeochemical cycles and occurs in a number of different oxidation states in natural and constructed treatment wetlands. Numerous biological and physiochemical processes can transform nitrogen between its various oxidation states (Figure 3-10). The dominant nitrogen species entering a FWS treatment wetlands depends on the level and type of wastewater pretreatment, but may include organic, ammonia, nitrate, and nitrite nitrogen, and nitrogen gases (di-nitrogen gas [N_2] and di-nitrogen oxide [N_20]).

A fraction of the organic nitrogen is readily mineralized to ammonia nitrogen in aquatic and wetland environments. Ammonia nitrogen is distributed between the ionized form (ammonium, NH_4^+) and a smaller percentage as un-ionized ammonia (NH_3). The distribution of total ammonia between NH_4^+ and NH_3 depends on water temperature and pH. Un-ionized ammonia is volatile and may be lost directly to the atmosphere.

Ammonium nitrogen can be oxidized in open, aerobic zones to nitrite and nitrate nitrogen through an aerobic microbial process called nitrification. Free dissolved oxygen and carbonate alkalinity are consumed in this process. Ammonium nitrogen may also be biologically assimilated and reduced back to organic nitrogen in the plants, or may be removed from the water column by adsorption to solid surfaces, such as wetland sediments. Adsorbed ammonium is readily released back to the dissolved ammonia state under anaerobic conditions.

Nitrate nitrogen is readily transformed to di-nitrogen gas in treatment wetlands by the microbiologically mediated anaerobic process, denitrification. Denitrification occurs most readily in wetland sediments and in the water column below fully vegetated growth where dissolved oxygen concentrations are low and available organic carbon is high. Organic carbon is consumed in this microbial process and alkalinity is produced.

FIGURE 3-10
Nitrogen transformation processes in wetlands (Gearheart 1998, unpublished data).

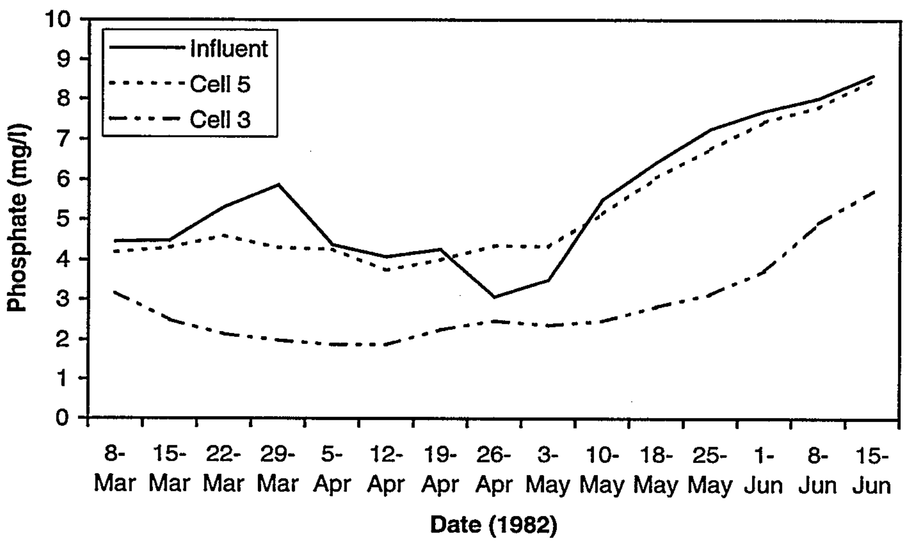


To complete the cycle, atmospheric di-nitrogen gas can be fixed by autotrophic organisms in open zones as organic N. However, this transformation is not normally a significant contribution of organic N to FWS treatment wetlands. Because of the complex transformations affecting nitrogen species in wetlands, a sequential series of reactions must be considered to adequately describe treatment performance, even on the most elementary level.

Phosphorus

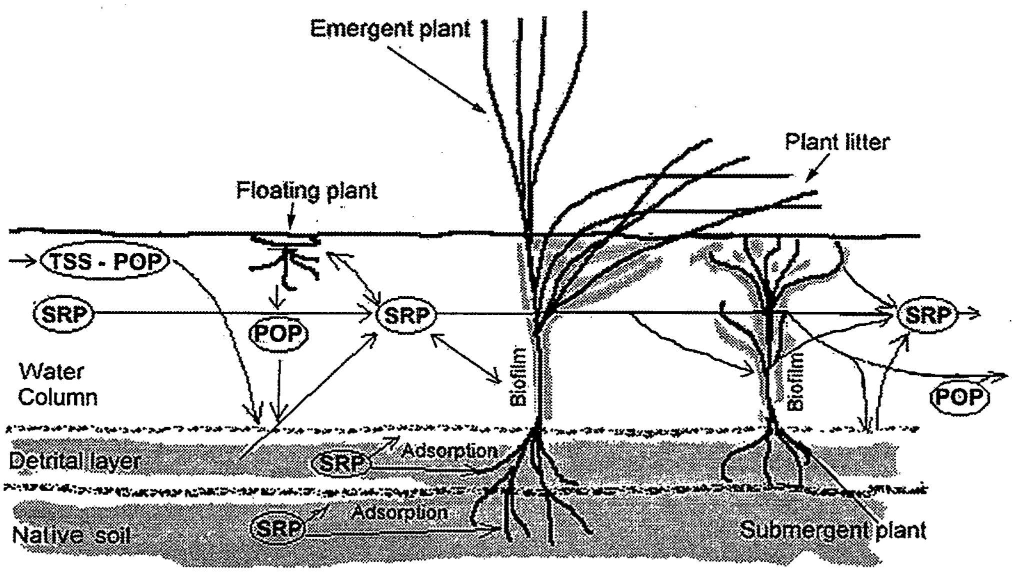
New constructed and natural wetlands are capable of adsorpting and absorbing phosphorus (P) loadings until the capacity of the soils and new plant growth are saturated. Phosphorus interacts strongly with the wetland soils and biota resulting in short-term removal and long-term storage (Reddy 1984, Reddy and D'Angelo 1994). The potential for P removal is most easily illustrated by the seasonal uptake and release by plants of soluble reactive phosphorus. The effects of two different phosphorus loadings on the effluent soluble reactive phosphorus during the growing season (Figure 3-11) were evaluated in Arcata's Pilot Project I. The difference between the lower phosphorus loading (Cell 3) and the saturated phosphorus loading (Cell 5) represents the mass of phosphorus taken up by macrophytes and epiphytes. The majority of the phosphorus taken up by the wetland plants is released as soluble reactive phosphorus in the late summer and fall as the vegetation senesce and decomposes.

FIGURE 3-11
Influent and effluent phosphorus in the Arcata Pilot Project I FWS wetlands, Second Pilot Project, 1982. Cell 5 was loaded at 0.75 kg/ha·d, and Cell 3 at 0.15 kg/ha·d (Gearheart 1993).



Sustainable P removal processes involve accretion and burial of phosphorus in wetland sediments. Uptake of P by small organisms, including bacteria, algae, and duckweed, act as a rapid-action, partly reversible removal mechanism (Figure 3-12). Cycling through growth, death, and decomposition returns most of the microbiotic uptake back to the water column, but a significant residual is lost to long-term accretion in newly formed sediments and soils. Macrophytes, such as cattails and bulrushes, perform a similar function, but on a longer time scale of months to years. The detrital residual from the macrophyte cycle also contributes to the long-term storage in accreted solids. Direct settling and trapping of particulate P may also contribute to the accretion process (Reckhow and Qian 1994). There can also be biological enhancement of mineralogical processes, such as iron and aluminum uptake and subsequent P binding in detritus and the algae-driven precipitation of P with calcium.

FIGURE 3-12
Conceptual cycling of phosphorus forms in FWS constructed wetlands. SRP: Soluble reactive phosphorus; POP: particulate organic phosphorus; TSS-POP: form of POP in terms of a fraction of the total suspended solids.



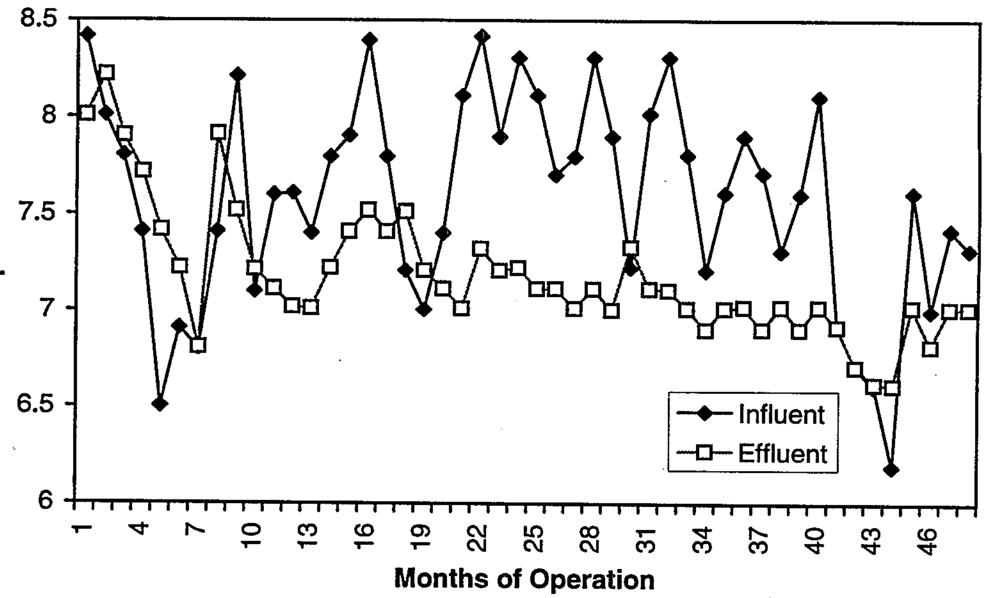
Hydrogen Ion

Natural wetlands exhibit pH values ranging from basic in prairie wetlands (8-9), to slightly basic in alkaline fens (pH = 7 to 8), and to quite acidic in sphagnum bogs (pH = 3 to 4) (Mitsch and Gosselink 1993). Natural freshwater marsh pH values are generally slightly acidic (pH = 6 to 7). Treatment wetland effluent hydrogen ion concentrations are typically neutral to slightly acidic. Data from an open water, unvegetated treatment "wetland," displayed high pH during some summer periods (pH > 9), with circumneutral influent (7.0 < pH < 7.4) (Bavor et al. 1988). Algal photosynthetic processes peak during the daytime hours, reducing dissolved CO_2 concentrations, creating high pH during the day, followed by a night-time sag with low pH as respiration replaces photosynthesis.

Organic substances generated within a wetland via growth, death, and decomposition cycles are a 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 are less well buffered against incoming acidic substances as the water column contains a limited amount of soluble humics.

The net result of the processes described above is that treatment wetlands can maintain their effluent pH at approximately pH 7 (Gearheart et al., 1983). Listowel, Ontario constructed treatment wetland No. 3 received lagoon water, which periodically exhibited high pH due to algal activity in the lagoon (Figure 3-13). During the first year of operation, little or no buffer capacity was evident. As the vegetation spread to cover the wetland and litter formation and decomposition became operative, high incoming pH values were neutralized effectively by the wetland.

FIGURE 3-13
Hydrogen ion (pH) buffering in system 3 at Listowel (Herskowitz 1986).

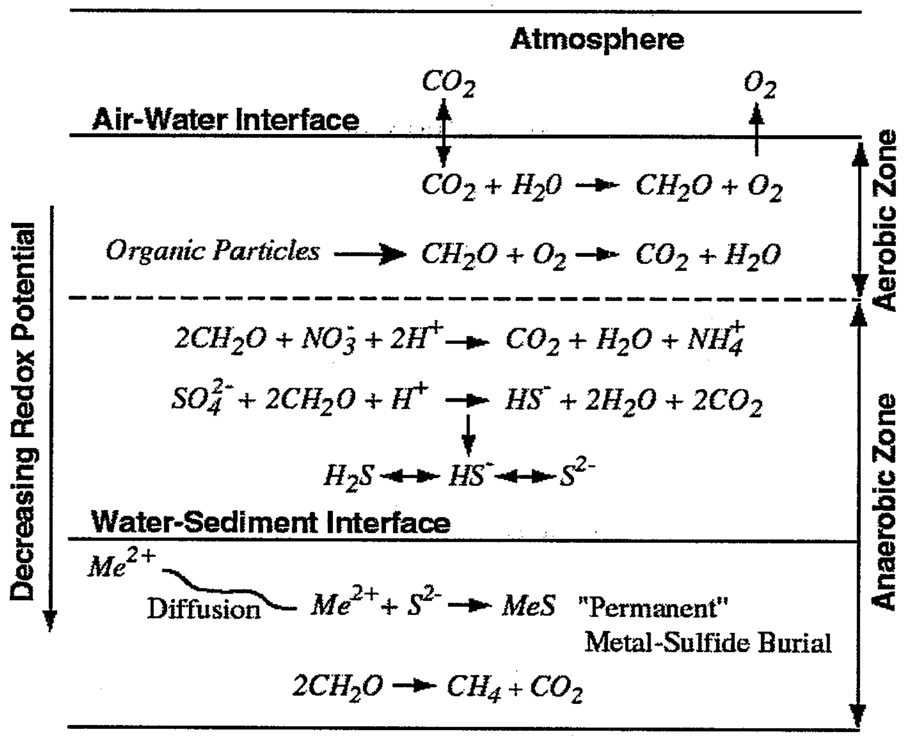


Metals

Metals removed from the water column by settling are bound to particles and may eventually be buried in the anoxic sediments. As shown in Figure 3-14, in the sediments below open water zones, many metals of concern are chemically reduced and bound as metal sulfides, a form that can minimize their biological mobility. If sediments are disturbed, the potential exists for the chemically reduced and sequestered metals to be oxidized and dissolve, thus becoming biologically mobile again. Metal actions in sediments below vegetated zones behave similarly, except that the aerobic zone is extremely shallow.

Metals are also incorporated into biomass via primary production processes occurring in a wetland. For macrophytes, metals are taken up via the roots and distributed throughout the plant. The extent of uptake and distribution within the plant depends on the metal species and plant type.

FIGURE 3-14
Metal sulfide burial processes in a wetland (Meyers 1998, personal communication).



Thermal Effects in Wetlands

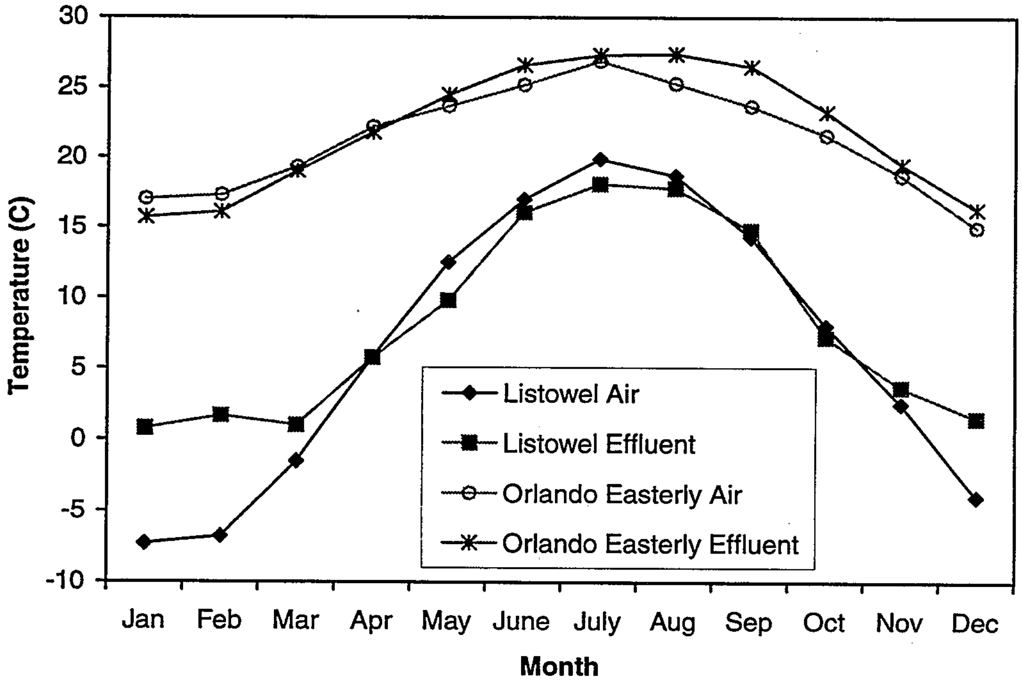
The temperature of wetland waters influences both the physical and biological processes within a FWS constructed wetland. Under winter conditions, ice formation may also alter wetland hydraulics and limit oxygen transfer. Under severe conditions, freezing may even result in system failure. Decreased temperatures are known to reduce the rates of biological reactions, the extent of which, however, varies with the constituent. In FWS constructed wetlands, BOD removal does not always appear to exhibit temperature dependence. Temperature dependent BOD removal may be masked by other processes, such as internal loads due to decomposition that are also temperature dependent, or the removal may be primarily due to non-biological mechanisms. Nitrogen removal has consistently been observed to decrease with temperature, indicating that it is controlled by biological mechanisms.

Predicting and understanding the influence of water temperature within a FWS wetland is an essential step in identifying the limits of its operation. Temperatures can be estimated using an energy balance that accounts for the gains and losses of energy to the wetland over time and space. The important gains and losses in the energy balance will vary seasonally. At minimum, a winter and summer energy balance will be needed to predict the range of operating water temperatures, and thus the corresponding range in temperature dependent pollutant removal rates.

In summer, large amounts of energy are supplied by solar radiation. A small portion of this recharges the soil energy storage, but most is lost via evaporation and transpiration. In winter, energy gains are from soil storage, and loss is to the cold ambient air. If snow or ice is present, radiation, convection, and sublimation create a balance that dictates the snow surface temperature.

When ice cover is absent, the energy balance is typically such that the gains and losses of energy are balanced, and the water temperature approaches equilibrium with the mean monthly air temperature, T (Figure 3-15).

FIGURE 3-15
Correlation between wetland water temperature and air temperatures. Both northern (Listowel) and southern (Orlando Easterly) systems show effluent water temperatures that follow the mean daily air temperature during warm months from nearby weather stations (Kadlec and Knight 1996).



If a frozen season is present, insulating layers of snow and ice can change the application of the energy balance considerably. There is no longer a large radiation input to the water, and energy gains are now solely from deep soil storage. Losses are by heat conduction through the snow and ice to the cold air above and to ice formation. Incoming sensible heat is typically dissipated because losses are generally greater than gains. Evaporation from the water layer is prevented by the ice cap. As a consequence, gains and losses do not balance as in summer, and temperature decline will typically proceed throughout the flow path.

The amount of ice formation is determined by climatological conditions that vary greatly from one winter to another. Wetland vegetation is effective in trapping snow to greater extents than unvegetated areas. Therefore, ice thickness in wetlands may be much less than in adjacent lakes or frost depths in nearby uplands. The Listowel, Ontario, wetlands experienced ice thickness of about 100 to 150 mm during flow conditions for a climate typified by a mean January air temperature of -9 °C (16 °F). Ice or frost depths in the Houghton Lake, Michigan, wetland range from zero (for copious early snow) to 200 mm for unvegetated pond zones with little snow. The mean January temperature is -8°C (18°F), and there is no winter water flow. Kadlec and Knight (1996) and Reed et al. (1995) provide a thorough discussion of FWS wetland temperature and ice formation prediction.

Constituent Characteristics

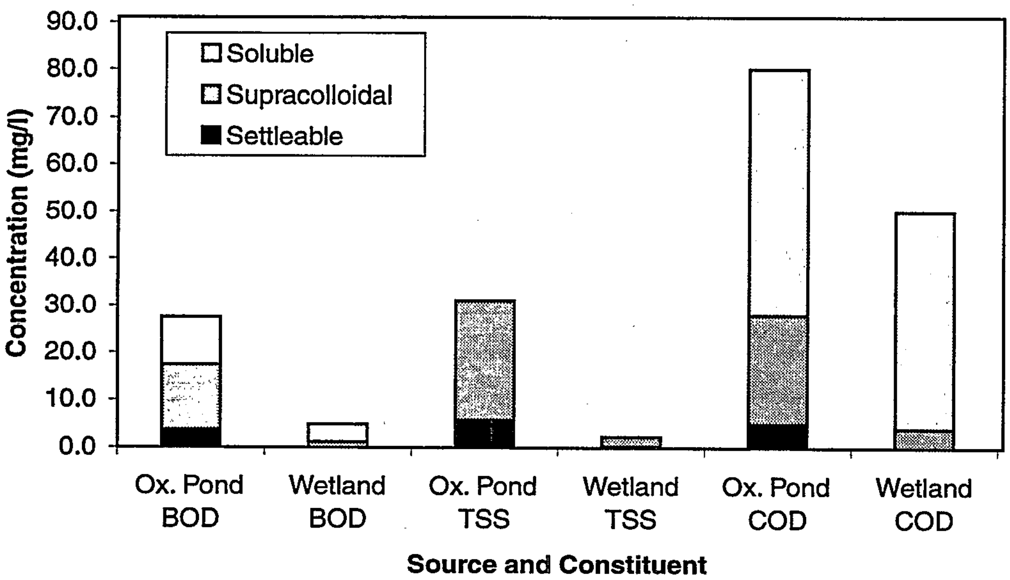
The characteristics (size, density, solid or dissolved), of wastewater constituents are of major importance in analyzing the performance of any wastewater treatment processes. These characteristics change as the wastewater proceeds through various processes in wastewater treatment systems. Wetland processes can play a role in the separation and solubilization of various wastewater constituents.

In effect, a FWS constructed wetland replicates a full wastewater treatment train in terms of types and linkages of the physical, chemical, and biological processes. It is important in the design and operation of a FWS wetland to determine the particulate/soluble distribution of the constituents. Settleable organic solids separated by settling process will serve as an internal load of dissolved and colloidal solids upon anaerobic decomposition. Biodegradable dissolved organic solids (VSS) are broken down, releasing ammonia, soluble reactive phosphorus, dissolved organic carbon, and gases (CO_2 and CH_4). Colloidal solids are also released in the decomposition process and include the heterotrophic bacteria responsible for the decomposition, as well as organisms and/or viruses of public health significance. The latter two particle types are adsorbed or affected in the settleable solids and are released to the water column upon decomposition.

As FWS constructed wetlands are placed further into conventional treatment trains; i.e., secondary and tertiary applications as opposed to primary and/or advance primary applications, the physical characteristics (size, density, etc.) of the constituents must be taken into consideration.

Soluble forms of COD and BOD dominate in the effluent from a wetland. An example of the partitioning of the various particle size of the constituents can be seen in Figure 3-16. In the case of an oxidation pond effluent, the majority of the BOD, for example, is supracolloidal. In the case of the wetland effluent, the majority of the BOD is soluble. It is the removal of the settleable and supracolloidal BOD through the wetland, which accounts for the majority of the BOD removed in the wetland. The soluble BOD removed is also significant and represents a net removal since the decay of the settled solids and plant detritus add to the soluble BOD in the system. This can be seen in the COD values in the oxidation pond and wetland effluent. The COD values are about the same for both systems. The BOD/COD ratio has changed significantly through the system as refractory and more complex organic compounds are formed in the decomposition of the plant material.

FIGURE 3-16
Distribution of BOD and COD concentration by form (settleable, supracolloidal, or soluble) in oxidation pond effluent and treatment marsh effluent from Arcata, California (Gearheart 1992).



Aquatic Vegetation

Types of Wetland Vegetation

Of the thousands of vascular plants on earth, only a relatively limited number are adapted to the conditions of continual submergence and waterlogged soils. FWS wetlands may consist of a variety of different emergent, submerged, and floating aquatic vegetation species, distributed primarily based on water depths. In general, emergent species are found in shallow water depths, while submerged species occupy deeper water zones; floating species of vegetation can occur in both shallow and deeper water areas.

In FWS constructed wetlands, the most common vegetation species have typically been emergent species such as bulrush, cattails, rushes, and reeds (Pullin and Hammer 1989, Reddy and Smith 1987). In the past, general practice was to use either a mono-culture of one species, or a combination of two or more species in FWS constructed wetlands used primarily for the treatment of wastewater. Constructed FWS wetlands that are used as habitat or enhancement wetlands, will typically be planted with a variety of emergent, submerged, and floating species. Some of the more common wetland plants used in FWS constructed wetlands, either for treatment or enhancement, and the species type and typical water depths of occurrence are listed in Table 3-2.

TABLE 3-2
Some common wetland plants and depths of occurrence used in FWS and floating aquatic constructed wetland.

Plant type	Species name	Common name	Range of depths (m)
Emergent	*Typha* spp.	Cattail	> 0.1 to < 1
	Scirpus spp.	Bulrush	> 0.1 to < 1
	Juncus spp.	Rushes	> 0.1 to < 0.3
	Carex spp.	Sedges	> 0.1 to < 0.3
	Phragmites spp.	Reeds	> 0.1 to < 1
Submerged	*Potamogeton* spp.	Pond weeds	> 0.5
	Vallisneria spp.	Tapegrass, wild celery	> 0.5
	Ruppia spp.	Widgeongrass	> 0.5
	Nuphar spp.	Spatterdock	> 0.5
	Elodea spp.	Waterweed	> 0.5
Floating	*Lemna* spp.	Duckweed	Flooded
	Eichhornia crassipes	Water hyacinth	Flooded
	Hydrocotlye umbellata	Water pennywort	Flooded
	Azolla spp.	Water fern	Flooded
	Wolffia spp.	Watermeal	Flooded

Vegetation Patterns

Treatment wetlands develop large amounts of emergent vegetation in areas with water depth less than about 60 cm deep. In general, larger nutrient supplies produce larger standing crops. These plants influence treatment performance in many ways, including:

- Uptake and cycling of nutrients and other elements
- Providing substrate for microbes and epiphytes, which process pollutants
- Creating drag on the flowing water, thereby creating head loss
- Occupying some of the water column, thus excluding liquid volume

- Increased plant biomass can increase the background concentrations of COD and BOD, which can amplify nutrient pulses from the effluent as a result of the seasonal decay of the vegetation in temperate climates with distinct growing seasons.

Above-ground macrophyte biomass may be separated into three compartments: live (green), standing dead (brown, upright), and litter (brown, broken, prostrate). Different compartments dominate the above-ground vegetation structure during different seasons. In northern climates, the end-of-season standing live crop converts to standing dead, and subsequently to litter. In warmer climates, such phases are shorter and less pronounced, but there are dormant periods at all latitudes.

Constructed wetlands do not initially possess all vegetative compartments; typically many months to a few years are required for the vegetative and litter compartments to fully develop (Figure: 3-17 and 3-18). In Figure 3-18, grass and duckweed that were predominant during the first year were relatively uncommon in the second year as cattail and hardstem bulrush grew taller and either shaded or filled in the open water areas. During this developmental period, carbon- and plant-dependent wetland functions may not be operating at their full potential.

FIGURE 3-17
Newly constructed wetlands require a startup period to attain full vegetative cover. Ground level and aerial reconnaissance were used to follow this process for the Tarrant County project (Alan Plummer Associates Inc. [APAI] 1995). The litter layer developed subsequently.

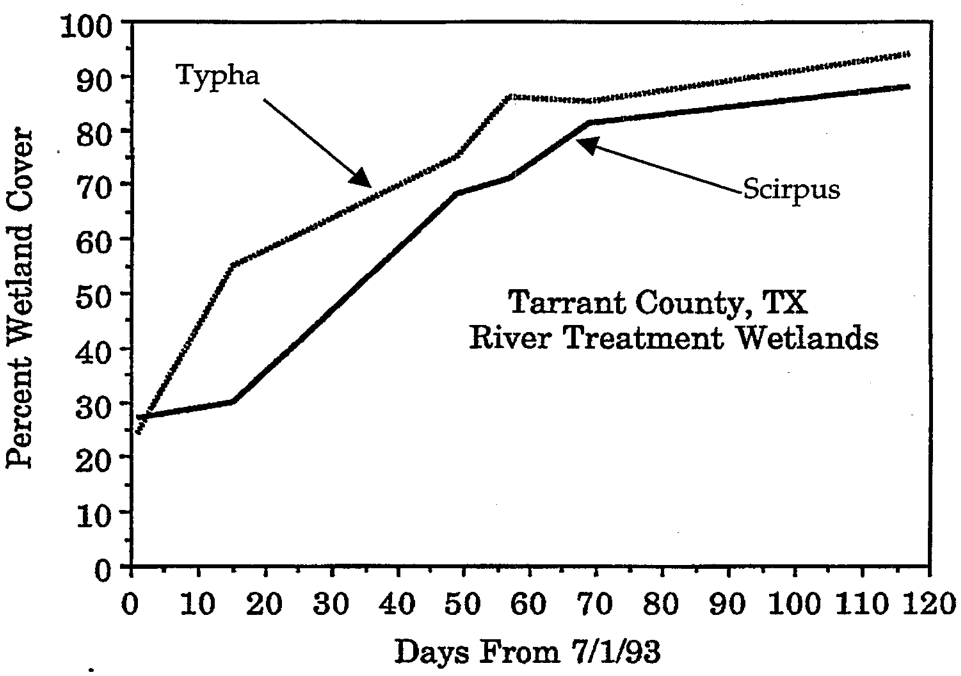
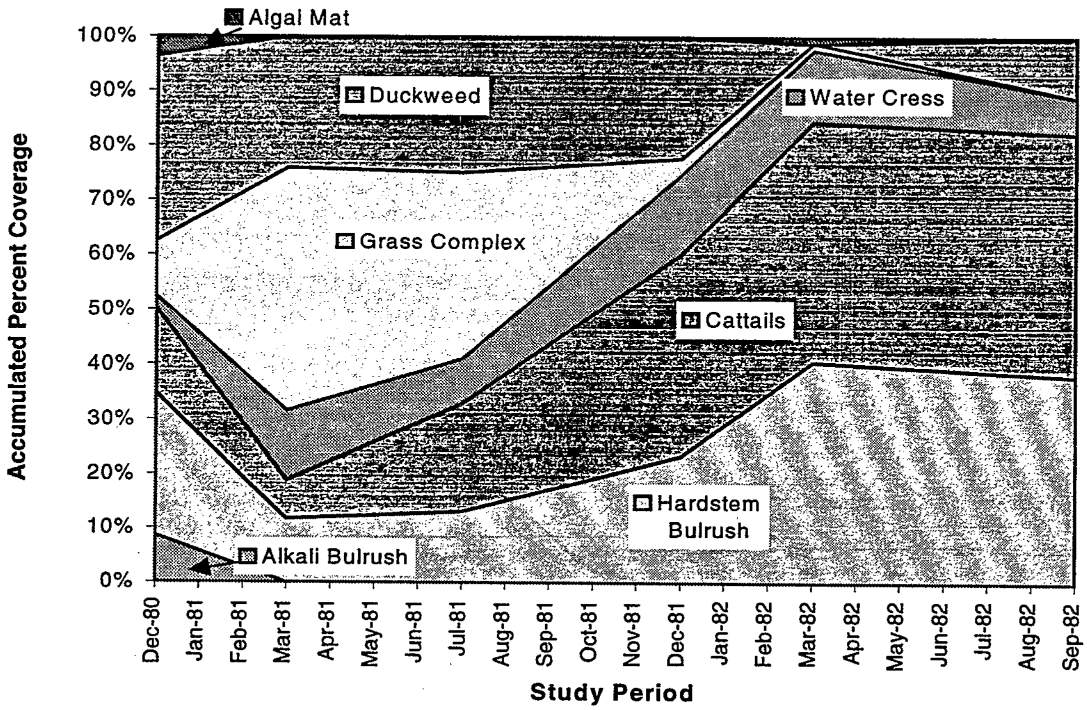


FIGURE 3-18
Coverage of plants during the startup period of the Arcata Pilot Project wetland.



The amount of biomass is climate and species-specific, as is the stem density. Cattail (*Typha* spp.) has a relatively large basal diameter, and occurs at about 40-50 stems per square meter in treatment wetlands. In contrast, bulrushes (*Scirpus* spp.) have smaller stems and may occur at hundreds per square meter. Stem density is additionally constrained by the growth requirements of the plant in question.

The space occupied by submerged plant parts acts to reduce detention time compared to an empty basin of the same depth. Plants block a small fraction (0 to 5 percent), and standing dead and litter can add a comparable fraction of blockage, leading to a total of 0 to 10 percent. In combination this can result in wetland porosities from 0.65 to 0.75 where the lower value is associated with dense mature vegetation (Reed et. al. 1995).

In contrast to the deleterious effect on detention time and head-loss, more submerged surfaces have the potential to house more microbes and epiphytes, and thus potentially enhances treatment. The amount of submerged area contributed by stems and leaves has been measured to range from 1.0 to 7.6 times the bottom area (Table 3-3). Dead plant parts are comparable in biomass and may contribute a comparable surface area, as does the un-vegetated wetland bottom in the absence of litter.

Role of Aquatic Plants in Controlling Treatment Processes

Aquatic macrophytes play an important role in the treatment processes active within FWS constructed wetlands. The plants, unique to the wetland environment, both influence the pollutant removal processes and act as sources and sinks of certain dissolved and particulate water quality constituents. Wetland plants also play an important role in preventing incoming radiation from entering the water column. Interception of incoming radiation significantly reduces algal growth, which can add carbon back to the system via photosynthesis. The shading of the water surface also can moderate the water temperature of a wetland. A distinguishing characteristic of FWS constructed wetlands is that the water temperature profile is buffered from abrupt changes in the ambient temperature. The cooling potential for any one site is dependent upon the range of temperatures found at that site, the ET rate, and the extent of the canopy. While the magnitude of thermal buffering is unique to a site, in certain locations this effect can be taken advantage of to meet receiving water temperature standards.

Well-developed stands of vegetation also reduce the natural reaeration process by influencing the micrometeorology within the wetland and limiting wind induced turbulent mixing. Lower rates of oxygen transfer, combined with low algal concentrations and the dissolved oxygen consumed within the water column to satisfy oxygen demands, usually results in low dissolved oxygen concentrations in FWS constructed wetlands. Surface level dissolved oxygen concentrations at 20 to 40 percent of saturation are commonly observed.

While debate surrounds the potential for in-situ reaeration via emergent macrophytes, no debate exists concerning the ability of submergent plants to contribute dissolved oxygen. In most cases, emergent and submergent plants are not found in the same wetland zones. Submergent aquatic macrophytes can thrive in un-shaded regions of FWS constructed wetlands. These plants contribute dissolved oxygen directly to the water column while affording a physical substrate for periphytic and epiphytic bacteria and algae. Plants such as *Potamogeton pectinatus* (sago pondweed), are commonly planted in FWS constructed wetlands to support the nitrification of ammonia and serve as a food source for aquatic waterfowl.

Floating aquatic macrophytes are subject to being moved by the wind over the surface of the open water. It is not uncommon to have plants such as *Lemna spp.* windrowed amongst and against emergents or a berm, resulting in nearly complete inhibition of normal photosynthetic reaeration processes. Proprietary processes have been developed to keep floating aquatic macrophytes from being redistributed by the wind through various anchoring mechanisms.

Wetland vegetation is also a source of dissolved and particulate material that combines with the influent wastewater to produce a mixture of biodegradable compounds similar to the production of BOD via algal growth and degradation in an oxidation pond. A broad range of heterotrophic and autotrophic organisms is capable of degrading these compounds.

Many of the biochemical transformations that occur in treatment wetlands are mediated by a variety of microbial species residing on solid surfaces such as those areas provided by plant leaves, stems, and litter. Examples of these processes include the decomposition of organic matter, periphyton fixation, nitrification-denitrification, and sulfate reduction. For example, maximum biofilm production of 1500 mg/m^2·d (dry-wt) has been measured in wastewater treatment wetlands at 60 percent of maximum sunlight (Tojimbara 1986). In turn, these processes are directly responsible for the water quality improvement potential of treatment wetlands. The submerged surface area of vegetation in a wetland is a function of plant type, plant density, and water depth. Reported submerged plant surface areas for various typical wetland plants are given in Table 3-3.

TABLE 3-3
Submerged surface area of wetland vegetation, normalized for a depth of 0.5 m.

Site	Dominant Vegetation	Submerged Area (m^2/m^2)	Depth (m)	Normalized Submerged Area (m^2/m^2)
Arcata, CA	*Scirpus acutis*	7.6	0.6	6.5
	Typha latifolia	2.6	0.6	2.2
Benton, KY	*Scirpus cyperinus*	1.8	0.25	3.6
	Typha latifolia	1.0	0.25	2.0
Houghton Lake, MI	*Carex* spp.	2.4	Unknown	Unknown
	Typha angustifolia	2.7	0.3	4.5
	Typha latifolia.	2.1	0.3	3.5
Pembroke, KY	*Scirpus validus*	1.2	0.2	3.0
	Typha angustifolia	1.5	0.2	3.7

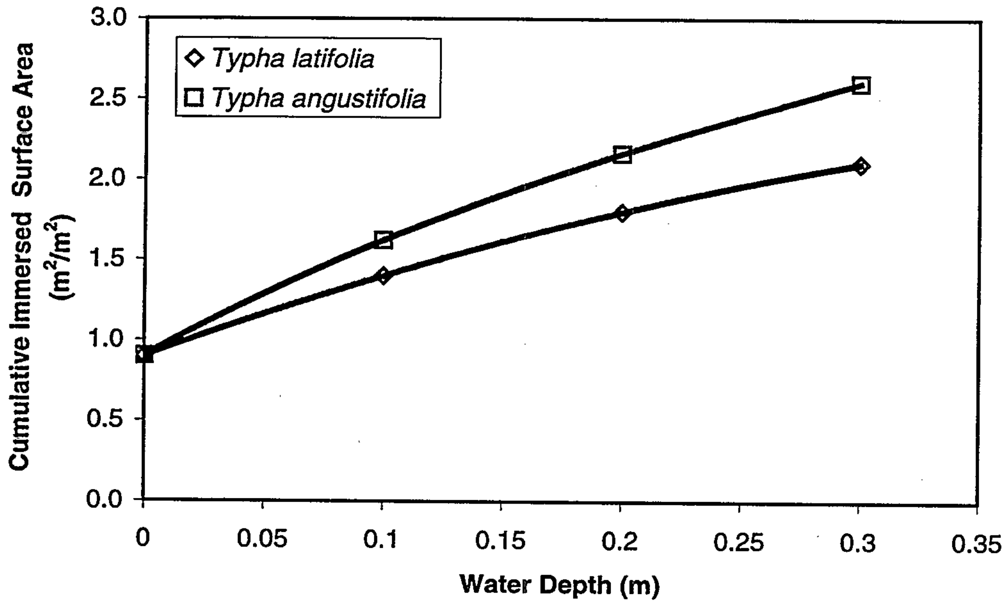
Source: Kadlec and Knight 1996, Kadlec 1997, Pullin and Hammer 1991, Gearheart et al. 1999 (publication in progress).

Depending on the dominant plant type, plant surface area may or may not be a function of wetland depth. If the primary contribution to surface area in the wetland is the bottom litter layer, then the surface area available for attached growth does not increase significantly with depth once the litter layer is submerged. In a scenario such as this, effluent quality may be largely independent of water depth (Kadlec and Knight 1996). For example, data from a sedge meadow at Houghton Lake indicated that very little additional surface area was observed for water depths greater than 250 mm (Kadlec, 1997).

In contrast, the surface area of live and dead plant material and litter for a *Typha* zone of the wetland at Houghton Lake is still showing a significant increase at

0.3 m (Figure 3-19). The change in leaf and stem (not litter) surface area with depth in a bulrush (*Scirpus acutis*) and cattail (*Typha latifolia*) zone of the Arcata Treatment Marsh are shown in Figure 3-20. In this example, the leaf and stem surface area continues to increase significantly up to the maximum depth measured of 1 meter. From these results, it can be concluded that in wetlands supporting plants that grow in deeper water (e.g., cattails and bulrush), the surface area for attached growth does increase with depth.

FIGURE 3-19
Stem, leaf and litter cumulative surface area for *Typha spp.* in Houghton Lake discharge zone wetland (Kadlec, 1997).

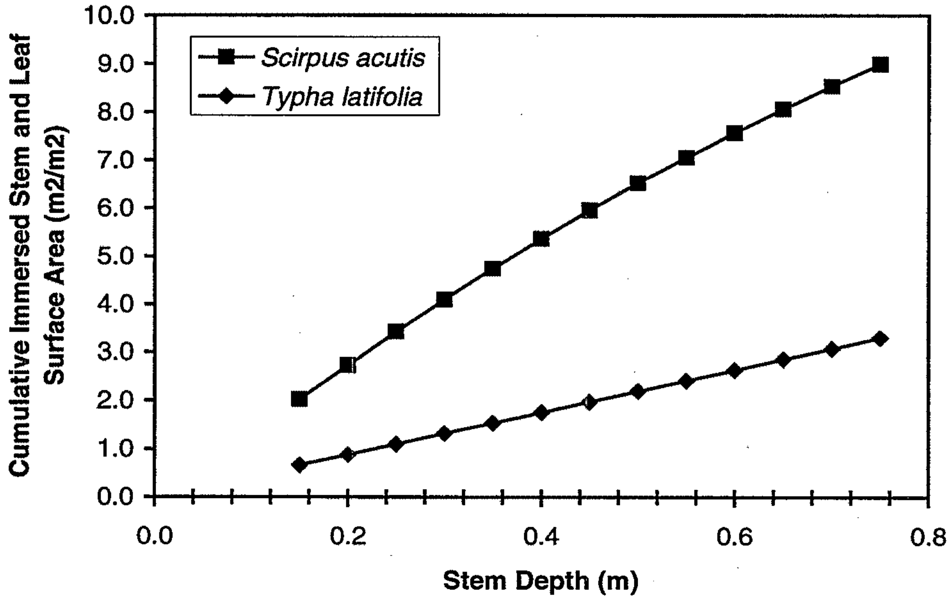


Wetland vegetation also has an effect on the hydraulic characteristics of the wetland, which directly influences water quality constituent removal processes. Wetland vegetation can

- Increase water losses through plant transpiration,
- Decrease evaporation water losses by shading water surfaces and cooling water temperatures,
- Create friction on the flowing water and, thereby, creating head-loss and flocculation of colloids,
- Provide wind blocks, thus promoting quiescent water conditions and protection for floating plants such as duckweed,
- Provide complex water column flow pathways, and
- Occupy a portion of the water column, thus decreasing detention time

In summary, it is the vegetation, specifically the emergent and submergent vegetation that gives a FWS constructed wetland its capability to treat wastewater effectively in a passive manner. Free water surface constructed wetlands are unique in that they grow their own physical substrate for periphytic microorganisms and epiphytic plants while minimizing incoming radiation addition. The fact that fixed-film biological reactions, sedimentation, and anaerobic digestion can all occur in an aquatic system can be attributed to the ecosystem created by the aquatic macrophytes. Without this vegetated component, the same physical conditions would result in an oxidation pond producing a large amount of total suspended solids (algae) in the effluent.

FIGURE 3-20
Stem and leaf surface area for *Scirpus acutis* (hardstem bulrush) and *Typha latifolia* (cattail) in Arcata Treatment Wetland (Gearheart et al., 1999, publication in progress).



SECTION 4

Performance Expectations

Approach to Performance Evaluation

Free water surface constructed wetlands tend to function as a sequence of coupled processes: discrete settling, flocculent settling, and benthic decomposition (ammonification and release of soluble degradable organics, soluble BOD removal, nitrification, phosphorus uptake, denitrification, etc.). The contribution and even presence of each process within a FWS constructed wetland is highly dependent on the design and operation of the treatment wetland.

The performance and permit compliance of operating FWS constructed wetland treatment systems reveals the range of effluent quality and the variability in performance, possible with these types of systems. Evident in this analysis is the range of conventional treatment strategies provided to wastewater, and thus the range of constituent loads, to which FWS treatment wetlands are subjected. Further, many of the wetlands systems with sufficient operational and design data represented in the NADB could be characterized as constructed wetlands receiving high quality effluent from advanced wastewater treatment processes. This allows *in-situ* contributions of BOD, TSS, and nutrients to dominate the wetland effluent and hence add to the variability of the wetland effluent quality. This section describes and compares the performance of a subset of operating FWS constructed wetlands for which sufficient data and information were available.

In addition to the performance assessment, an analysis of permit compliance for those sites that had both permit limits and operational data of comparable frequency available in the NADB is included. Actual system operational flows were compared to design flows as a measure of the loading to each system during the period evaluated. For sites with adequate data, the length of the data record, the percent compliance, and the average and maximum effluent concentrations during that period are reported. Because of limitations of the NADB, a subset of the operational data from most of these systems is included in this analysis.

The performance assessment and permit compliance analyses presented are organized by constituent, with subsections for BOD, TSS, nitrogen, phosphorus, fecal coliform, metals, and organics. The chapter concludes with a discussion of background concentrations and stochastic variability.

Methodology of Performance Evaluation

The FWS constructed wetlands used in this technology assessment were the systems which best met the minimum criteria for inclusion for analysis (see Section 2 and Table 2-5). This Technology Assessment Database (TADB) includes selected systems from the NADB and additional systems for which operational data are available. The wetland sites with sufficient water quality and operational data to be included in the TADB and used for this technology assessment are reported in Table 4-1. For some systems, data on all water quality constituents were available while for other systems only select constituents were available. Permit compliance was evaluated using systems in the NADB, but not necessarily in the TADB.

The performance evaluation of FWS constructed wetlands has been analyzed at three different levels. The first level includes a summary analysis of all available data for systems listed in Table 4-1, calculation of the mean influent and effluent concentrations and their range of values. The mean and range of mass loadings for each water quality constituent are given in Table 4-2. This first level of assessment is useful in the context of summarizing the range of operating conditions of FWS constructed wetlands and the range of response in terms of effluent concentration. At this level, the wide range of applications and expected performance for operating FWS treatment wetlands are summarized. No accounting for differences in upstream waste treatment processes, geometric configuration, planting strategy, inlet/outlet works, climate, etc. has been made even though, each of these factors can significantly affect the effluent quality of a FWS constructed wetland.

In the second level of performance data analysis, those systems with the most extensive monthly influent/effluent data for the constituents of interest are compared. This level of analysis is presented in terms of cumulative probability over the data collection period. The third level of analysis is designed to determine how individual systems perform in terms of effluent concentrations over the range of their loadings. In the third level of analysis, monthly loadings versus effluent concentrations for a single site are compared, thus demonstrating the expected variability within a single system.

TABLE 4-1
Water quality constituent data availability for the FWS constructed wetland systems included in this assessment (TADB), as identified in Table 2-6.

Wetland System	Water Quality Parameter									
	BOD	TSS	NH_4-N	TKN	NO_3-N	TN	OrgN	TP	DP	FC
Arcata Pilot I Cell 8	•	•	•							•
Arcata Pilot II	•	•	•		•					•
Arcata Treatment	•	•								
Arcata Enhancement Allen	•	•								
Arcata Enhancement	•	•						•		
Beaumont	•	•	•							
Benton Cattail	•	•	•	•	•	•	•	•	•	•
Benton Woolgrass	•	•	•	•	•	•	•	•	•	•
Brookhaven Meadow Marsh	•	•	•	•	•	•		•	•	•
Cannon Beach	•	•	•					•	•	•
Central Slough	•	•	•	•	•	•	•	•		•
Clermont Plot H			•		•	•	•	•	•	
Columbia	•									
Fort Deposit	•	•	•	•	•	•	•			
Gustine (89-90) 1A	•	•	•	•	•					
Gustine (89-90) 1B	•	•	•	•	•					
Gustine (89-90) 1C	•	•	•	•	•					
Gustine (89-90) 1D	•	•	•	•	•					
Gustine (89-90) 2A	•	•	•	•	•					
Gustine (89-90) 2B	•	•	•							
Gustine (89-90) 6D	•	•	•	•	•					
Gustine (94-97)	•	•								
Houghton Lake			•		•					
Iron Bridge	•	•	•	•	•	•	•	•	•	•
Lakeland	•	•				•				
Listowel 4	•	•	•	•	•	•	•	•	•	•
Manila	•	•								
Minot	•	•	•							
Mt. Angel	•	•	•							
Orange County	•	•	•	•	•	•	•	•	•	•
Ouray	•									
Pembroke FWS 2	•	•	•	•	•	•	•	•	•	•
Poinciana Boot	•	•	•	•	•	•	•	•	•	•
Reedy Creek WTS1	•	•	•	•	•	•	•	•	•	
Reedy Creek OFWTS	•	•	•	•	•	•		•	•	
Sacramento	•		•	•	•	•		•		
Sea Pines Boggy Gut	•	•	•	•	•	•	•	•	•	•
Tres Rios Hayfield	•	•	•	•	•			•		
Vereen Bear Bay	•	•	•	•	•	•	•	•		•
West Jackson County	•	•	•	•	•	•	•	•		

TABLE 4-2
Summary of performance data and loadings for TADB systems analyzed in this assessment (listed in Table 4-1).

	Influent (kg/ha·d)			Influent (mg/L)			Effluent (mg/L)		
Constituent	Min	Mean	Max	Min	Mean	Max	Min	Mean	Max
Biological Oxygen Demand (BOD)	0.04	31	183	1.7	70	438	1.2	15	69
Total Suspended Solids (TSS)	0.07	22	92	1.0	69	588	1.1	15	40
Ammonia (NH_4-N)	0.02	3.5	16	0.63	8.7	29	0.07	6.8	23
Total Kjeldahl Nitrogen (TKN)	0.04	5.8	20	1.3	18	51	0.82	11	32
Nitrate (NO_3-N)	0.05	0.9	3.5	0.31	3	13	0.01	1.2	3.5
Total Nitrogen (TN)	0.12	3.0	9.9	2.1	12	32	0.85	4.0	9.8
Organic Nitrogen (OrgN)	0.02	1.8	5.7	0.74	5.6	18	0.71	2.1	3.2
Total Phosphorus (TP)	0.01	1.2	4.4	0.27	4.1	11	0.09	2	4.2
Dissolved Phosphorus (DP)	0.01	0.6	1.3	0.23	2.6	5.7	0.04	1.5	3.7
Fecal Coliform (FC) (col/100mL)				1.7	73,000	360,000	47	1,320	9,800

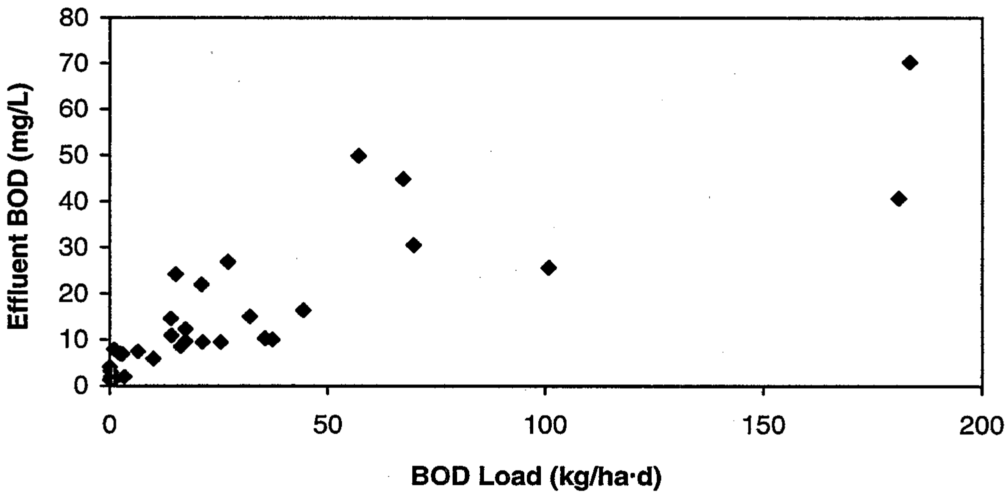
BOD Performance

Database Assessment

The relationship between average BOD loading and average BOD effluent concentration for TADB systems shown in Figure 4-1. There is a general linear trend between increased BOD loading and increased effluent concentration over the loading range of 0.1 to 180 kg/ha·d. Considering the wide range of conditions, wetland design, and data quality, a general trend exists between increased loading and decreasing effluent quality. Specific systems have BOD effluent versus BOD loading curves, which are better correlated and predict lower effluent quality compared to the general trend observed.

As shown in Figure 4-1, considerable effluent variation exists for a given BOD loading. At a BOD loading of 25-kg/ha·d, the effluent concentrations vary from 9 to 35 mg/L. At lower BOD loading rates, the effluent BOD varied from 1 to 8 mg/L (BOD loading rate of 0.1 to 8 kg/ha·d). The effect of the background BOD due to plant decomposition is evident in systems with low loading rates. In addition to plant decomposition, relatively small changes in the inlet/outlet region, levels of animal activities, or weir location and operations, can all significantly affect the effluent BOD concentration under low loading rates.

FIGURE 4-1
Average BOD loading rate versus effluent BOD concentration for TADB sites.



Temporal BOD Performance

A summary of BOD loading versus effluent BOD concentration for the treatment and enhancement wetlands at Arcata, California, is given in Figures 4-2 and 4-3, respectively. Seven years of monthly data shows the normal variation observed in two systems in the same location receiving different loads. As seen in Figure 4-2, the treatment marsh effluent BOD concentration is sensitive to influent BOD while the enhancement marsh effluent is not as sensitive to the influent BOD concentration.

FIGURE 4-2
Monthly influent and effluent BOD values for Arcata's treatment wetland.

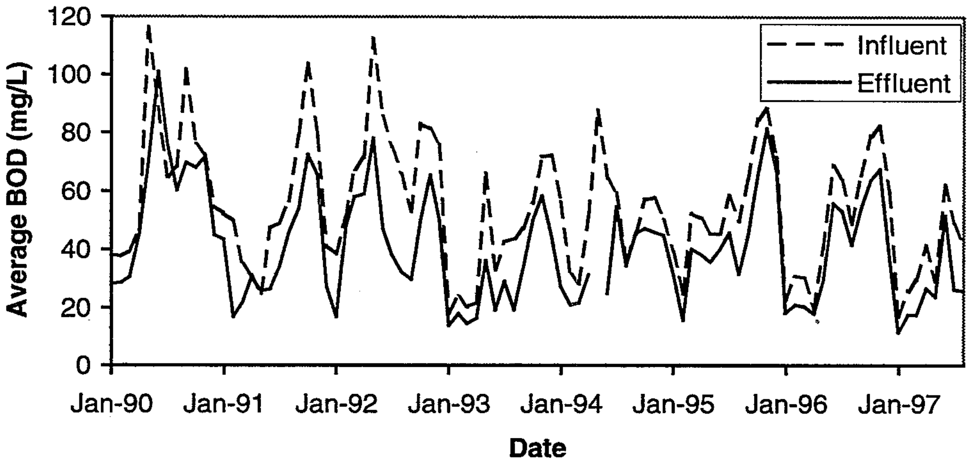
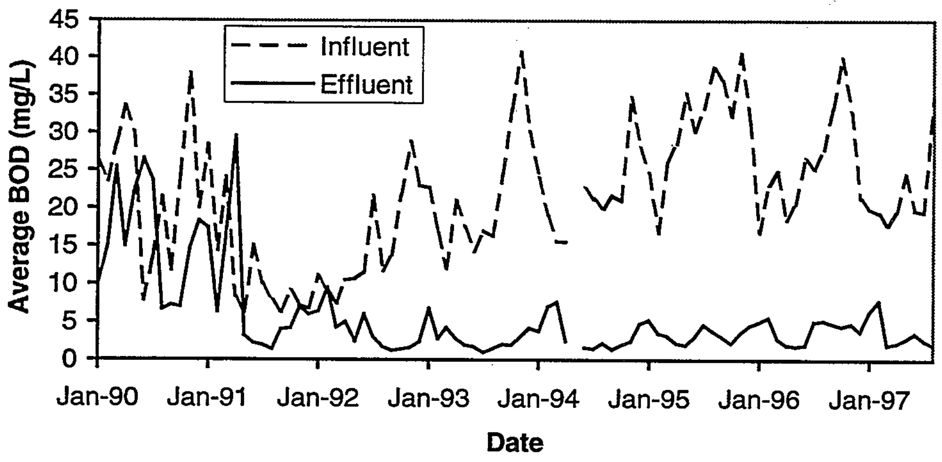
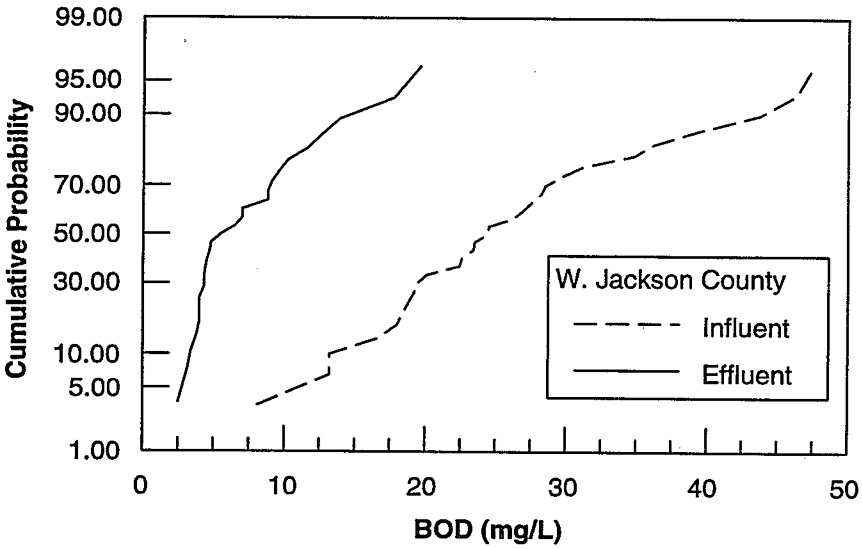


FIGURE 4-3
Monthly influent and effluent BOD values for Arcata's enhancement wetland.



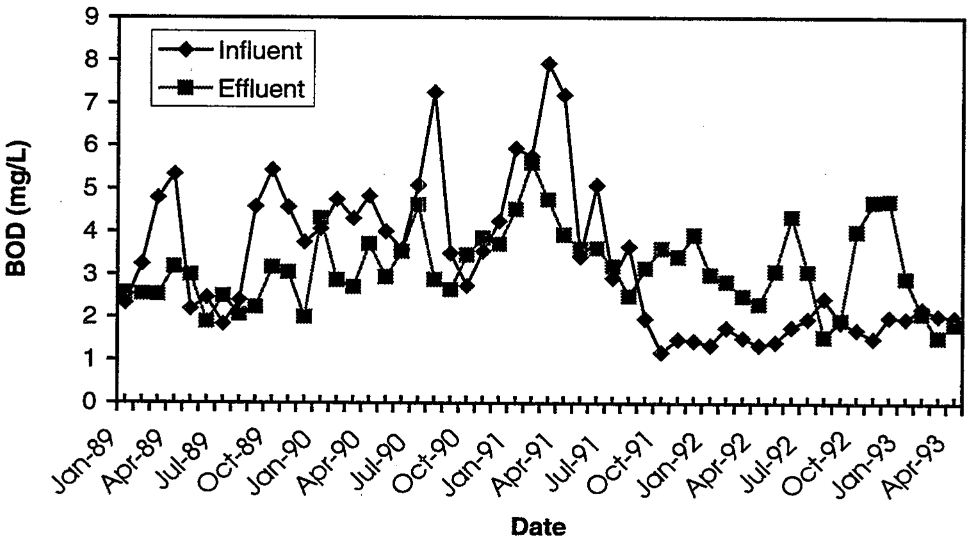
Effluent cumulative probability BOD levels from West Jackson County, Mississippi, are shown in Figure 4-4. This particular system shows effluent concentrations between 2 and 20 mg/L over influent BOD levels ranging from 8 to 48 mg/L with a mean effluent BOD of 4 mg/L and a mean influent value of 24 mg/L.

FIGURE 4-4
Influent and effluent monthly BOD cumulative probability values for West Jackson County, Mississippi.



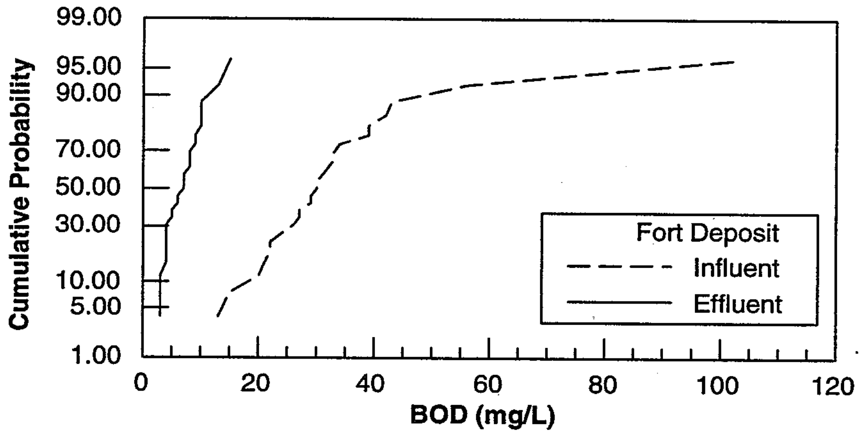
Influent and effluent BOD data for Lakeland, Florida, are given in Figure 4-5. In this system, the majority of influent values are less than 5 mg/L and during one 12-month period, the effluent BOD is greater than the influent. In this case, the internal processes producing total suspended solids and dissolved BOD from aquatic plant and epiphytic primary production and decomposition increase the effluent BOD above the influent BOD.

FIGURE 4-5
Influent and effluent monthly BOD for Lakeland, Florida.



The Fort Deposit influent and effluent BOD data are presented in Figure 4-6. As shown in the figure, this system exhibits consistently effective BOD removal. Effluent BOD concentrations are almost always low, between 2 and 15 mg/L, while the influent concentration varies from 18 to 100 mg/L.

FIGURE 4-6
Influent and effluent monthly BOD cumulative probability for Fort Deposit, Alabama.



The relationship of BOD loading to effluent BOD concentration for the Arcata Treatment Marsh is shown in Figure 4-7. The BOD loading ranged from 76 to 605 kg/ha·d, with an average of 180 kg/ha·d. As might be expected, better relationships between loading and effluent concentrations were found on a site-by-site basis than observed by lumping data from all the sites together or even when comparing two systems at the same site.

The Arcata Treatment Marshes have removed BOD at a constant rate of 68,000 kg/ha·yr, for the last 7 years. These three treatment wetlands with a total area of 1.86 ha operate in parallel and remove approximately 30 percent of their influent BOD. This constant removal rate can be seen in Figure 4-8, in which the accumulated BOD mass in and out of the treatment wetland is plotted.

Effluent BOD concentration from the Arcata Pilot Project can be predicted using Equation 4-1:

$$C_e = 3.42 + 0.262\, C_i \qquad (4\text{-}1)$$

Where: C_e = effluent BOD (mg/L)

C_i = influent BOD (mg/L)

This equation fit the 3 years of experimental data for cells with hydraulic residence times ranging from 6 to 12 days, with an R^2 of 0.91, indicating a fairly constant relationship.

FIGURE 4-7
Monthly BOD loading rate versus BOD effluent concentration for Arcata Treatment Marsh.

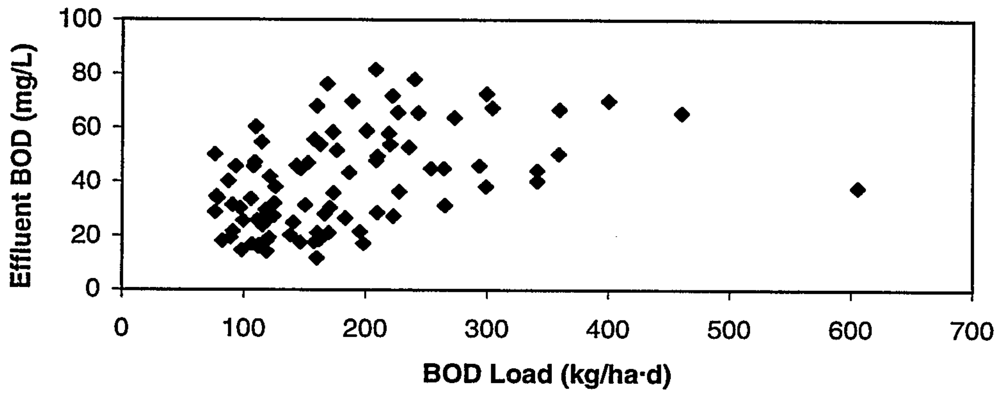
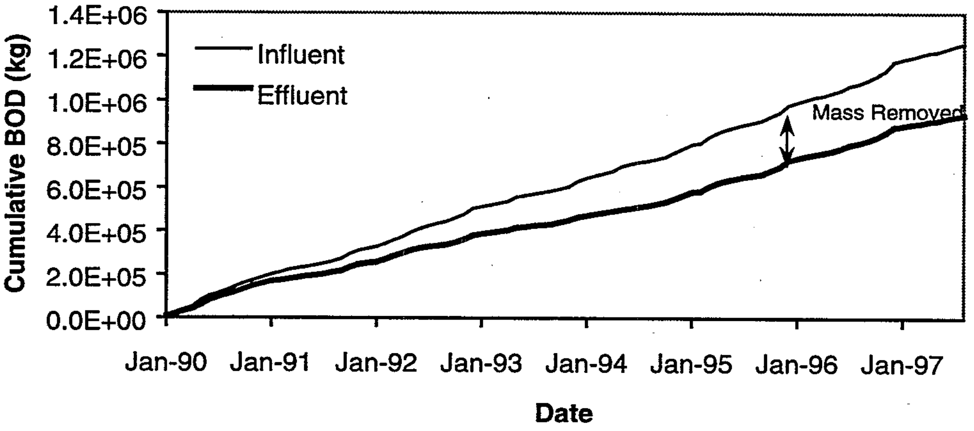


FIGURE 4-8
Cumulative monthly mass influent and effluent BOD for the Arcata Treatment Wetland. The area between the two curves is representative of the mass of BOD removed.



BOD Permit Compliance

Enough information was available from the NADB to evaluate BOD_5 permit compliance for 12 FWS treatment wetland systems. Effluent BOD_5 permit limits varied from 5 to 30 mg/L on a monthly average basis. Summer BOD_5 limits were more restrictive than winter limits for five of these systems. In general, FWS constructed wetlands have been very effective at meeting BOD_5 effluent limits, even as low as 5 mg/L.

Only four of the 12 FWS constructed wetlands had less than 100 percent compliance with BOD_5 permit limits during the analyzed period of record. The Central Slough, South Carolina, natural treatment wetland exceeded its effluent permit limit of 30 mg/L just once in 69 months of operational data. Flow to this system was about 40 percent of design capacity during that period. The Fort Deposit, Alabama, constructed treatment wetland exceeded the summer BOD_5 limit of 10 mg/L one month out of seven with a concentration of 13 mg/L. Flow at that time was only about 54 percent of design capacity. The Norwalk, Iowa, system exceeded its BOD_5 limit of 30 mg/L six times during the 35-month record analyzed. The maximum-recorded effluent BOD_5 during this period was 70 mg/L. Flow averaged about 58 percent of design flow during that period. The Pembroke, Kentucky, constructed wetland exceeded its 10 mg/L limit about 67 percent of the time during a 9-month period. The maximum, recorded effluent value was 24 mg/L at an average flow of 84 percent of design.

TSS Performance

Database Assessment

The effectiveness of FWS treatment wetlands to remove TSS is recognized as one of their principal advantages. The relationship between TSS loading and effluent TSS levels for the entire data set is shown in Figure 4-9. Over a range of loadings from 0.5 to 180 kg/ha·d, there does not appear to be any relationship between loading and effluent quality with this data set. What is apparent is that under a fairly wide range of solids loadings, relatively low effluent TSS concentrations can be attained.

Because physical processes dominate the removal of TSS, it is expected that, to a point, TSS effluent levels are not affected by hydraulic or solids loading rates. The dominant TSS removal processes occur within the first 1 to 2 day hydraulic residence time (HRT) period. This effect can only be seen in transect data with 1 to 2 day increments. Most of the wetlands in the wetland database have detention times in excess of 2 days, which allows the removal of TSS to be masked by subsequent internal generation of TSS. The variation in the effluent TSS shown in Figure 4-9 is most likely related to internal TSS sources such as algal growth, sloughed epiphytes, animal sources, re-suspension, or detrital particles.

In the case of TSS effluent cumulative probability distribution, there are examples of systems that are consistently effective and systems in which the background levels are sometimes greater than the influent. For example, the Fort Deposit, Alabama, influent TSS levels varied from 18 to 183 mg/L, with an average loading of 7.4 kg/ha·d, while the effluent TSS levels varied from 3 to 39 mg/L, representing a significant TSS removal rate (Figure 4-10). In contrast, Orange County, Florida, influent TSS ranged from 1 to 4 mg/L, while the effluent ranged from 1 to 17 mg/L, with an average effluent of 4 mg/L, 2.6 mg/L greater than the average influent concentration. Based on data from sites like Orange County, it can be concluded that wetlands generally will not reduce TSS concentrations below 3 mg/L.

Removal of TSS is most pronounced in the inlet region of a FWS constructed wetland. Transect data from pilot project studies at Arcata show this pattern of removal (Figure 4-11). Generally 50-60 percent of the TSS from oxidation pond systems is removed in the first 2-3 days of nominal hydraulic detention time. Gravity settling processes account for most of this removal, and the overall removal efficiency is a function of the terminal settling velocity of the influent biosolids. Within the TSS loading range of 50 to 200 kg/ha·d, the removal of the settled total suspended solids does not require any routine solids handling operation. The separated solids undergo anaerobic decomposition, releasing soluble dissolved organic compounds and gaseous by-products, carbon dioxide, and methane gas, to the water column.

Long term studies from individual sites have shown low and stable effluent concentrations from a relatively wide range of TSS loading rates. The TSS

FIGURE 4-9
Monthly TSS loading versus effluent TSS concentration for TADB wetland systems.

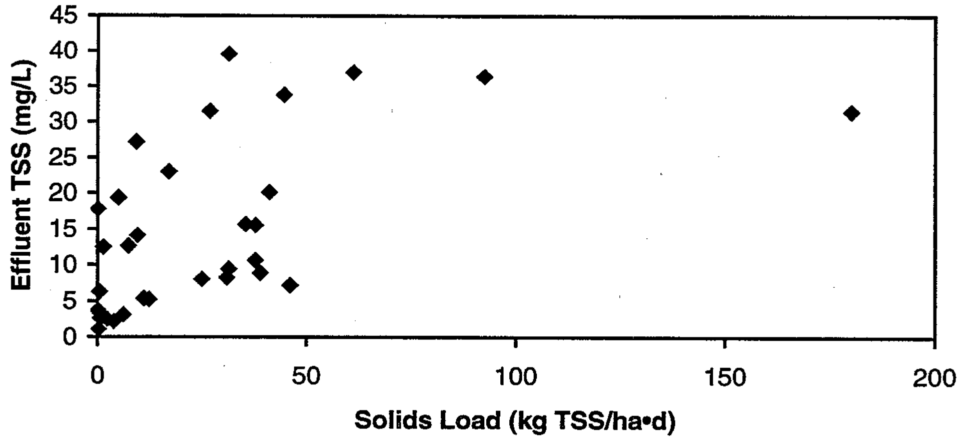
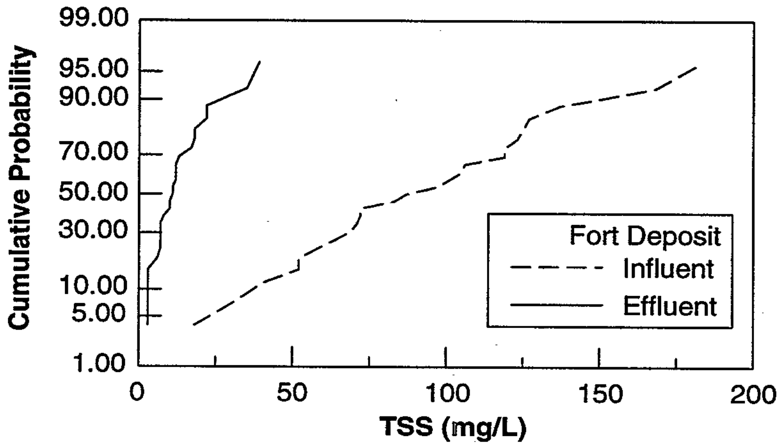


FIGURE 4-10
Cumulative probability distribution of monthly influent and effluent TSS concentration for Fort Deposit wetland.



effluent concentrations rates from the Arcata Enhancement Wetland are consistently low, less than 5 mg/L, 90 percent of the time, with an annual average loading of 16 TSS kg/ha·d (Figure 4-12). The Arcata enhancement marsh has continued to remove TSS at a constant rate of approximately 90 percent for the last 6 years. An operational change in January of 1991 increased the BOD removal rate, while TSS removal has remained constant. An increase in hydroperiod (depth increase from 0.25 to 0.5 m) coupled with no alteration in the outlet weir setting over the year has stabilized the effluent TSS and BOD levels. The effluent TSS concentration has not tracked the influent levels with the operational strategies used over the last 6 years.

FIGURE 4-11
Weekly transect TSS concentration for Arcata's Cell 8 Pilot Project, with theoretical retention time of 6 days, receiving oxidation pond effluent.

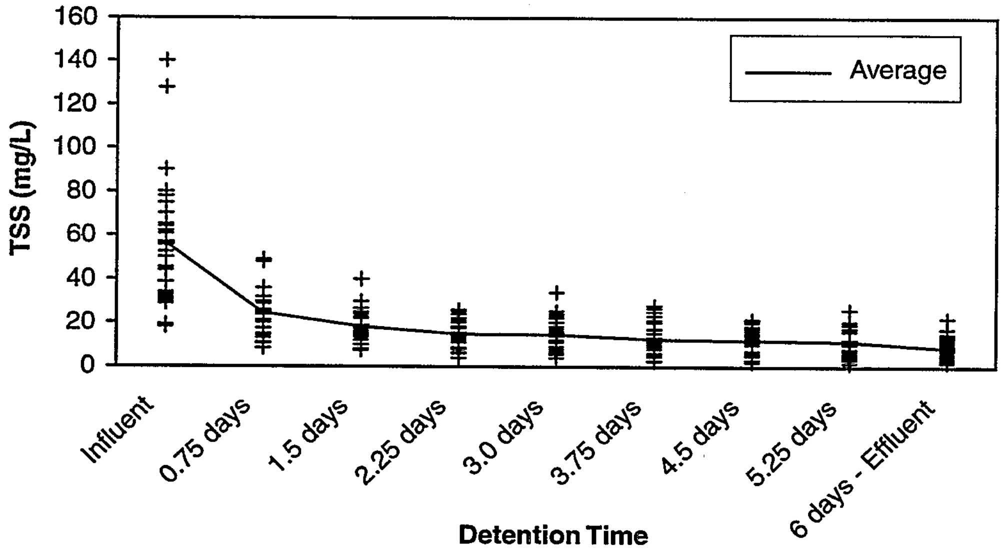
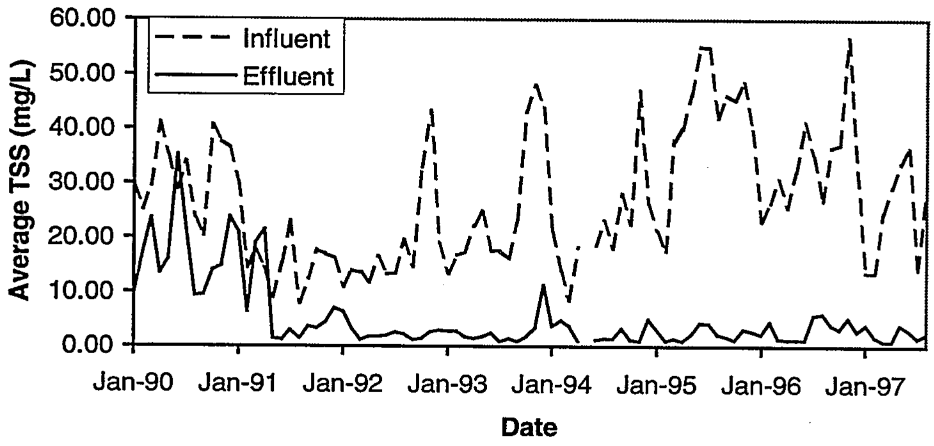


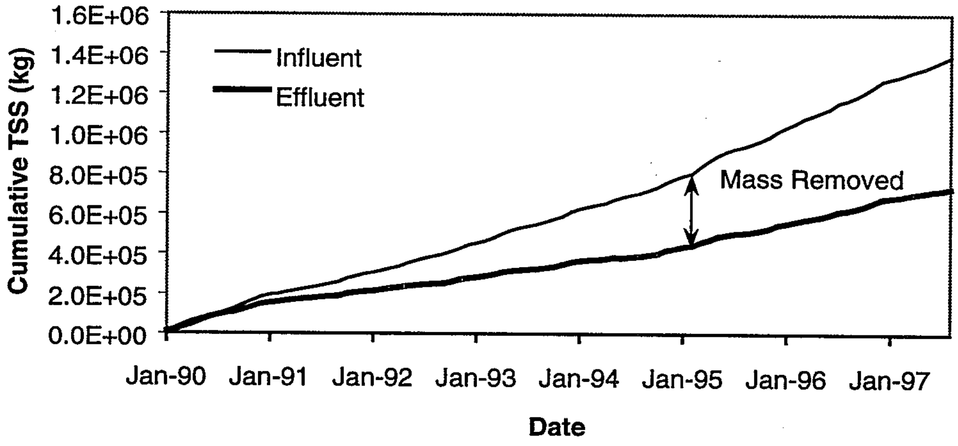
FIGURE 4-12
Weekly Influent and effluent TSS concentration for Arcata Enhancement Wetland.



Temporal TSS Performance

The Arcata Treatment Marshes have the highest average TSS loading in the TADB (180 kg/ha·d, average influent TSS of 60 mg/L), yet the removal has continued at a more or less constant rate of about 50 percent over the last 6 years (Figure 4-13). The TSS effluent levels from the treatment marsh are less than 27 mg/L, 50 percent of the time.

FIGURE 4-13
Cumulative yearly mass influent and effluent TSS for Arcata Treatment Wetland.



TSS Permit Compliance

Thirteen FWS constructed wetland systems with permit and effluent data were available in the NADB that could be used to evaluate permit compliance. Effluent TSS permit limits varied from 10 to 30 mg/L on a monthly average basis. One system (Reedy Creek, Florida) also had an annual average TSS limit. Only one of these systems had seasonal limits for TSS (Vermontville, Michigan).

In general, the FWS constructed wetlands were able to meet effluent TSS limits. The cases where limits were exceeded resulted from poor vegetative cover and the subsequent growth of phytoplankton or solids re-suspension. Of the 13 systems in the NADB, 8 had 100 percent compliance with TSS effluent limits.

Five FWS constructed wetlands had less than 100 percent compliance with TSS permit limits during the period of record in the NADB. The Central Slough, South Carolina, natural wetland exceeded a 30 mg/L effluent limit twice during 24 months of operational data, and had a monthly maximum of 66 mg/L during this period. Benton, Kentucky, Cell 2 exceeded its 30 mg/L permit level twice during 20 months, with a maximum during this period of 53 mg/L. Average flow to this cell was about 65 percent of design flow. Benton Cell 1 exceeded its permit limit of 30 mg/L three times during the same 20-month period. Average flow in this cell was also about 65 percent of design. The Norwalk, Iowa, constructed wetland was in compliance with the 80 mg/L permit limit about 69 percent of the time during the 35 months of record.

Nitrogen Performance

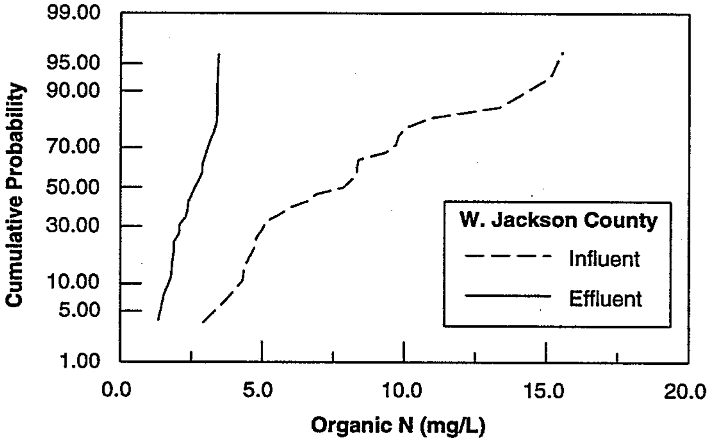
Effluent concentration data for nitrogen species shows considerable variation in response to the nitrogen loading. Total nitrogen (the sum of all nitrogen species) and total Kjeldahl nitrogen (organic plus ammonia nitrogen) effluent concentrations are generally correlated to their respective loadings. However, individual forms of nitrogen, ammonia, nitrate, and organic nitrogen, may exhibit very little correlation between effluent concentrations and influent loadings. This latter set of nitrogen species has both sources and sinks within FWS wetlands and a speciated nitrogen balance for a specific system is necessary to analyze removal performance.

In a number of cases, effluent concentrations of ammonia or nitrate N have been found to be higher than influent concentrations. This concentration increase is rarely the case for organic or total N. The conclusion from these observations is that the sequential nitrogen transformation processes result in an overall uni-directional conversion of elevated total and organic nitrogen forms to oxidized or gaseous nitrogen forms in treatment wetlands. However, these processes can also lead to increasing concentrations of intermediate nitrogen forms due to temporal and spatial differences in conditions necessary to support denitrification (alkalinity/carbon concentrations, and redox potential). Distribution of various species of nitrogen within a wetland indicates that the nitrogen dynamics are affected by the influent loading, the degree of plant coverage and maturity of emergent vegetation (Sartorius et al. 1999).

Organic Nitrogen Performance

Nearly all the FWS treatment wetlands that have been studied have reported reductions in total nitrogen and organic nitrogen. The transient nature of organic nitrogen is a consequence of the balance of sources and sinks active at a given site. Organic nitrogen is produced by anaerobic degradation and is converted to ammonia nitrogen by ammonification processes making it difficult to determine the relationship between organic nitrogen loading and effluent concentration. Analysis of performance data requires a complete nitrogen balance for a particular site; it is somewhat meaningless to use data from different sites. A better relationship between influent and effluent organic nitrogen was found for individual sites. For example, a consistent removal of organic nitrogen from influent mean values of 25 mg/L to effluent mean value of 8 mg/L is shown in the data from West Jackson County (Figure 4-14).

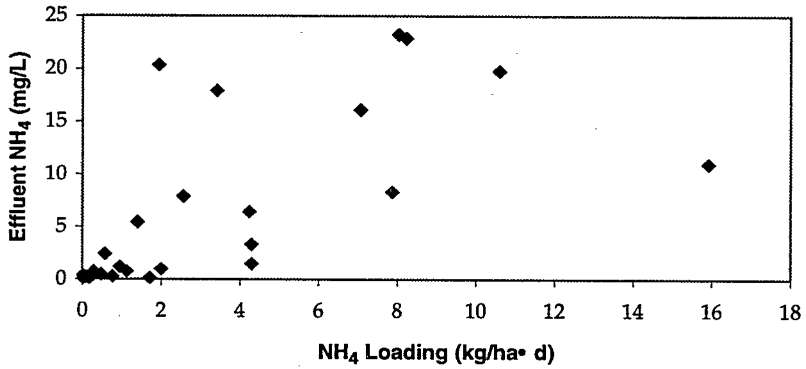
FIGURE 4-14
Cumulative probability distribution of influent and effluent organic nitrogen for West Jackson County, Mississippi.



Ammonia Nitrogen Performance

The ammonia effluent concentrations observed for the range of loading rates in the TADB is shown in Figure 4-15. Ammonia nitrogen effluent concentrations are poorly correlated with ammonia loading rates, due to the internal ammonia contribution from organic nitrogen (org N) associated with the TSS. Ammonia nitrogen shows considerable variability for a given loading. At loadings between 2.0 and 3.0 kg/ha·d, effluent ammonia concentrations ranged from 0 to 20 mg/L. Systems represented in the lightly loaded region generally showed low effluent ammonia levels.

FIGURE 4-15
Ammonia nitrogen loading versus effluent ammonia concentrations for TADB systems.



Presentation of ammonia loading versus effluent concentration data for a number of different systems tends to mask the relationship between the various forms of nitrogen, the influent concentrations of ammonia, the water temperature, and the detention time of the wetland. The Beaumont, Texas, FWS constructed wetland is an example of a system that showed very consistent ammonia nitrogen removal (Figure 4-16). Over a 4-year period, the 8 cell system of the Beaumont wetland had an average hydraulic detention time of 17.4 days, an average water temperature of 22.5 °C, and an average ammonia loading of 4.3 kg/ha·d. As shown in Figure 4-17, the average ammonia removal was nearly 90 percent.

Ammonia nitrogen levels in constructed wetlands can increase within the wetland as decomposing particles become soluble. This increase mirrors the contribution of dissolved organic carbon as settled solids decompose in the inlet zone of the wetland.

FIGURE 4-16
Cumulative probability distribution of monthly influent and effluent ammonia nitrogen from Beaumont, Texas.

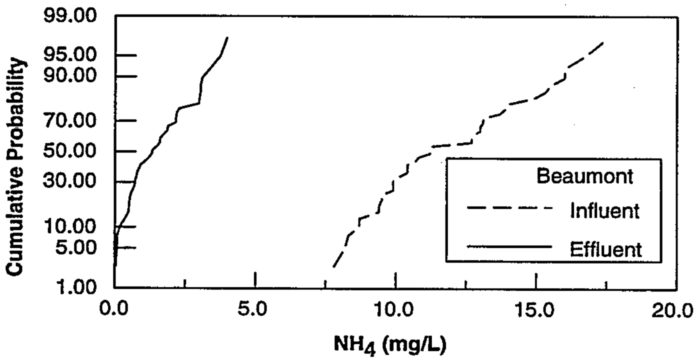
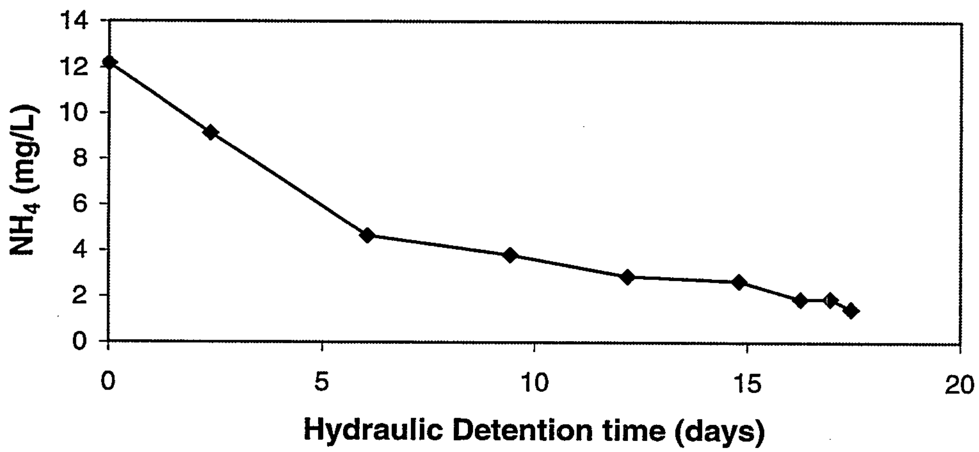


FIGURE 4-17
Ammonia nitrogen removal for Beaumont, Texas, through 8 cells with a total HRT of 17 days.



Total Kjeldahl Nitrogen Performance

Total Kjeldahl nitrogen (TKN) loading versus effluent levels for TADB systems shows general trends of increased loading producing increased effluent concentrations (Figure 4-18). Because TKN is the sum of the organic nitrogen and the ammonia, the correlation between influent and effluent TKN is expected to be higher than for the individual components because analyzing TKN eliminates the effects of internal conversion reactions between the organic and ammonia nitrogen. Generally, those systems with an influent TKN concentration less than 2 mg/L had effluent ammonia concentration significantly less than 1 mg/L, indicating that in treatment wetlands, the background level of TKN is attributable to the organic nitrogen. The cumulative probability distribution of the influent and effluent TKN concentration for the Central Slough wetland is shown in Figure 4-19. The Central Slough system had an average influent concentration higher than the TADB average (17 versus 12 mg/L), and an average removal rate of 75 percent, slightly higher than the TADB average of 67 percent.

FIGURE 4-18
Total Kjeldahl nitrogen loading versus effluent ammonia concentrations for the TADB.

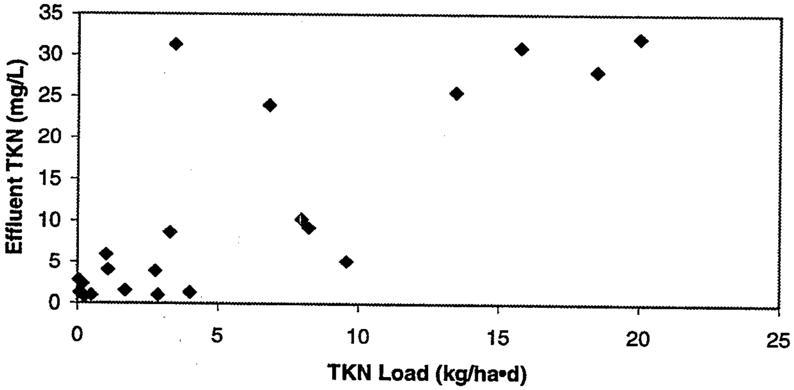
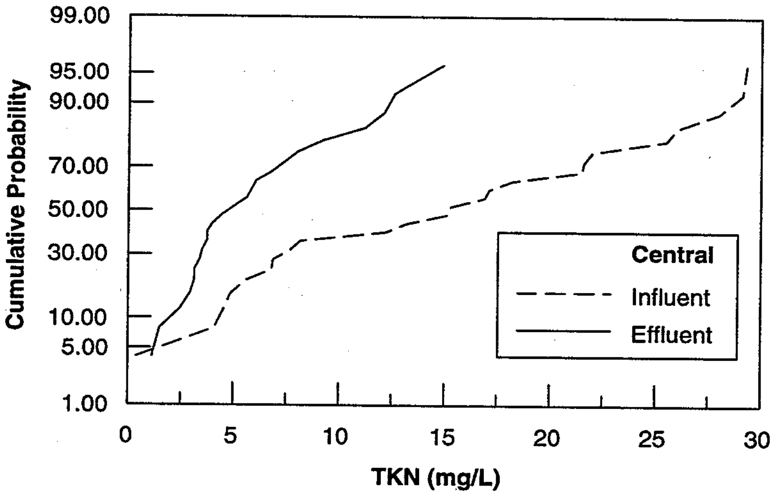


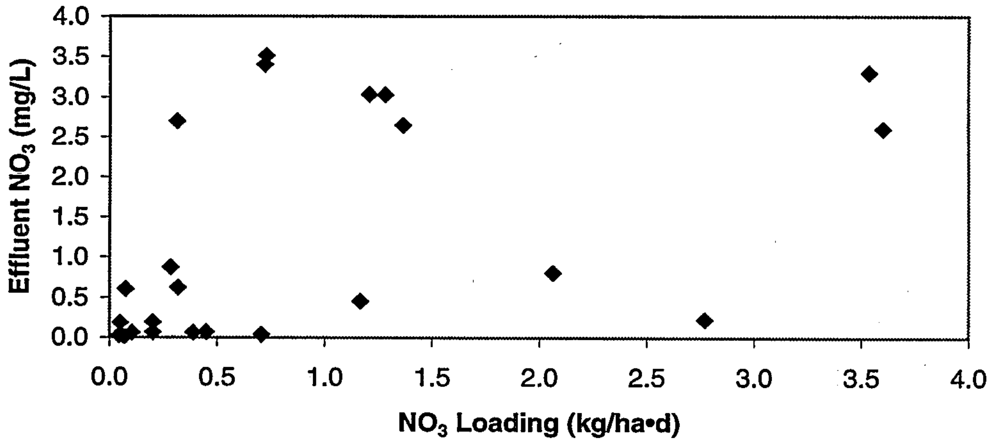
FIGURE 4-19
Cumulative probability distribution of monthly influent and effluent TKN from Central Slough, South Carolina.



Nitrate and TIN Performance

Nitrates are also transient nitrogen species in FWS wetlands. The extent of nitrate removal or production depends on the presence and distribution of aerobic (nitrification produces nitrate from ammonia) and anoxic (denitrification in which nitrate is converted to nitrogen gas) regions within a FWS wetland. As shown in Figure 4-20, essentially no relationship exists between nitrate loading and effluent quality in the TADB systems. Only in the case of a highly nitrified effluent would one expect to see a relationship between nitrate loading and effluent nitrate concentration.

FIGURE 4-20
Nitrate nitrogen loading versus effluent nitrate concentrations for the TADB.



The performance for Orange County, Florida, as shown in Figure 4-21 is typical of a lightly loaded system. The Orange County system has nitrate effluent concentrations less than 0.1 mg/L with mean influent nitrate concentrations of 0.80 mg/L. Iron Bridge operates under similar conditions with comparable performance, 95 percent of the effluent nitrate concentrations are less than 0.1 mg/L with a mean influent concentration of 1.1 mg/L.

The Arcata Enhancement Wetland receives a high loading of total inorganic nitrogen (TIN) (sum of nitrite, nitrate and ammonia nitrogen) and shows a TIN reduction from a mean of 26 mg/L in the influent to a mean of 4 mg/L in the effluent (Figure 4-22). Performance of this system was very consistent. Organic N is approximately 15 percent of the total nitrogen, while the majority (95 percent) of the TN is in the form of ammonia and nitrate nitrogen.

FIGURE 4-21
Cumulative probability distribution of monthly influent and effluent nitrate concentrations for Orange County, Florida.

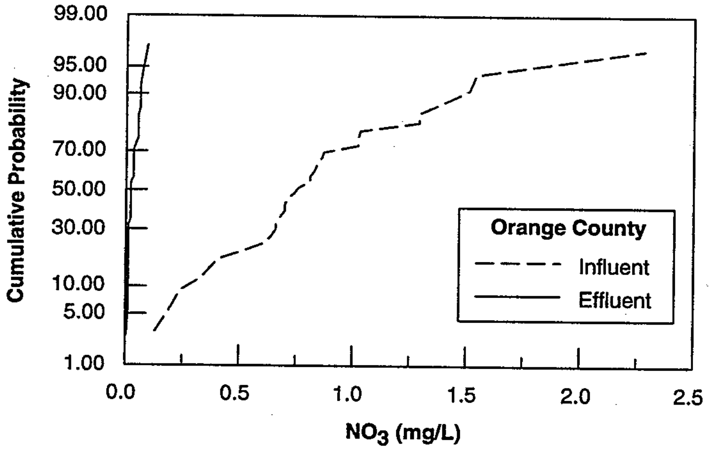
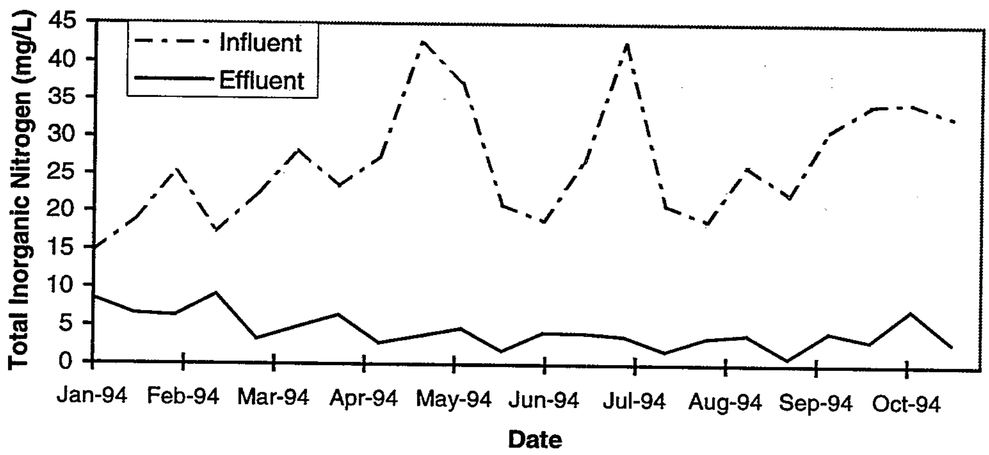


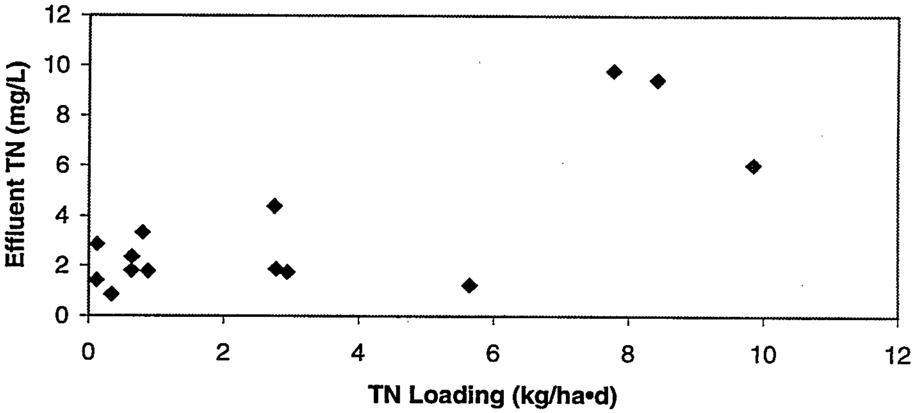
FIGURE 4-22
Monthly influent and effluent of total inorganic nitrogen (TIN) for the Arcata Enhancement Wetland.



Total Nitrogen Performance

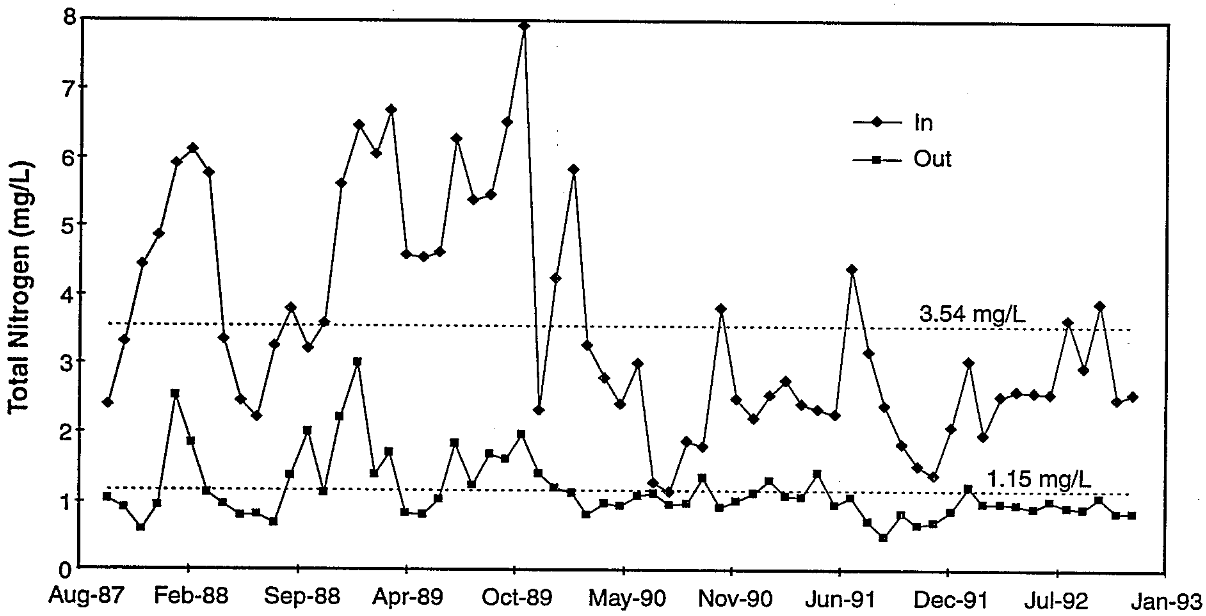
Total nitrogen, the sum of the organic and inorganic forms, in FWS constructed wetlands shows a correlation between increased loading and increased effluent concentrations (Figure 4-23). However, within the range of 0.1-6 3 kg/ha·d considerable variation exists in the effluent concentrations.

FIGURE 4-23
Total nitrogen loading versus effluent total nitrogen concentrations for TADB wetland systems.



The typical range of inlet and outlet TN concentrations for the first 12 cells of the FWS constructed wetland at Iron Bridge, Florida, is illustrated in Figure 4-24. Individual maximum monthly outlet concentrations are more than two times higher than the long-term average.

FIGURE 4-24
Range of monthly inlet and outlet TN concentrations for cells 1 through 12 at the Iron Bridge FWS wetland near Orlando, Florida.



Nitrogen Permit Compliance

Ammonia Nitrogen

Ten FWS constructed wetland systems with NH_4-N permit and effluent data were available for evaluation from the NADB. The NH_4-N effluent permit limits varied from 1 to 20 mg/L on a monthly average basis. Six out of ten of these systems had seasonal limits for NH_4-N.

Effluent NH_4-N limit compliance continues to be a challenge for FWS constructed wetlands. Of the ten systems in the NADB, only four had 100 percent compliance with NH_4-N effluent limits.

Six FWS constructed wetlands had less than 100 percent compliance with NH_4-N permit limits during the period of record in the NADB. Benton Cells 1 and 2 had 100 percent compliance with winter NH_4-N effluent limits of 10 mg/L, but they exceeded summer limits of 4 mg/L 83 percent of the time during their 6 months of record. Maximum outlet NH_4-N concentrations were about 12.5 mg/L at an average flow of approximately 69 percent of design flow. The Fort Deposit, Alabama, constructed wetland exceeded its NH_4-N effluent limit of 2 mg/L only once out of 25 months with a maximum monthly value of 4.84 mg/L. The Norwalk, Iowa, wetland exceeded its summer limit of 8 mg/L only one time out of 20 months of record in the NADB. The maximum monthly value was 16.3 mg/L for Norwalk. The West Jackson County, Mississippi, system missed its NH_4-N permit limit of 2 mg/L 6 months out of 33 months of record with a maximum value of 3.92 mg/L. Average flow during this period was about 96 percent of the design flow.

Total Nitrogen

Only four FWS constructed wetlands had TN permit limits and associated data in the NADB. The permit limits for TN varied from 2.0 to 2.5 mg/L for these wetlands. The Reedy Creek, Florida, natural wetland systems had annual average limits in addition to monthly limits. A few treatment wetlands receiving highly pretreated (fully nitrified) wastewater have been able to attain low TN effluent limits.

Two out of four systems in the NADB had 100 percent compliance with their TN effluent limits. The Iron Bridge, Florida, constructed treatment wetland met its TN effluent limit of 2.3 mg/L during all of the 63 months of record in the NADB at an average flow about 61 percent of design. Maximum TN outlet concentration recorded during this period was only 1.7 mg/L. The Orange County, Florida, hybrid treatment wetland (both constructed and natural cells in series) met a TN permit limit of 2.2 mg/L 86 percent of the 37 months of record. The maximum recorded TN value during this period was 2.6 mg/L at an average flow of about 48 percent of design. The Reedy Creek System 1 exceeded TN effluent permit limits of 2 to 2.5 mg/L about 15 percent of the time during the period reported in the NADB. The maximum recorded annual average TN outflow value for this system was 8.2 mg/L and was the result of a 6-month upset in the activated sludge conventional treatment system preceding the natural wetland.

Total Phosphorus Performance

Database Assessment

Total phosphorus removal in wetlands has been of great interest to system operators and researchers, thus the amount of data and analysis is much greater than for many other constituents. There are hundreds of wetland-years of performance data for phosphorus, spanning two decades. The majority of these studies focused on non-domestic wastewater phosphorus sources. While comparisons can be made, it is important to separate the inorganic particulate phosphorus performance from the organic particulate phosphorus performance.

Because of the great amount of study conducted regarding phosphorous in treatment wetlands, Table 4-3 is provided to illustrate the range of hydraulic loading rates and TP concentrations and resulting outlet concentrations (annual averages) for natural and constructed wetlands in the NADB. For the NADB sites considered the average TP annual average removal ranged from as low as 9.7 percent to greater than 98 percent. Overall, the mean average annual removal rate for this collection of sites was 61 percent with a standard deviation of 30 percent.

TABLE 4-3
Total Phosphorus Removal Rates for Non-Forested Treatment Wetlands (NADB, 1993).

Site	No. of Wetlands	Data Years	HLR cm/day	TP In mg/L	TP Out mg/L	TP Removal %
Hidden Lake, Florida	1	3	0.59	0.100	0.045	55.0
Des Plaines, Illinois	4	7	4.55	0.106	0.022	79.2
ENR, Florida	4	1	2.75	0.125	0.025	80.0
OCESA, Florida	4	6	0.97	0.212	0.042	80.2
Iron Bridge, Florida	5	8	1.21	0.252	0.069	72.6
Cobalt, Ontario	1	2	7.71	1.678	0.774	53.9
Listowel, Ontario	5	4	2.41	1.909	0.717	62.4
Great Meadows, Massachusetts	1	1	0.95	1.996	0.507	74.6
Houghton Lake, Michigan	1	18	0.44	2.983	0.100	96.6
Pembroke, Kentucky	2	2	0.77	3.015	0.115	96.2
Sea Pines, South Carolina	1	8	20.20	3.940	3.360	14.7
Fontanges, Quebec	1	2	5.60	4.150	2.400	42.2
Benton, Kentucky	2	2	4.72	4.540	4.098	9.7
Leaf River, Mississippi	3	5	11.68	5.167	3.964	23.3
Lakeland, Florida	7	7	7.43	6.540	5.690	13.0
Clermont, Florida	1	3	1.37	9.140	0.150	98.4
Brookhaven, New York	1	3	1.50	11.075	2.325	79.0
					Site Average	60.7±30.2

Source: NADB 1993

The relationship between the total P loading and effluent concentration for the TADB data set is shown in Figure 4-25. Over a range of loading from 0.5 to 4.5 kg/ha·d, total phosphorus effluent concentration increases with loading. At lower loading rates (<0.5 kg/ha-day), however, the effluent phosphorus concentration ranged from 0.1 to 1.5 mg/L. Mean site specific data from Central Slough for influent and effluent total phosphorus were 4.5 and 2.2 mg/L, respectively (Figure 4-26).

Thirteen TADB and 39 NADB sites reported dissolved phosphorous data that were grouped into four categories based upon the analytical method used; (1) orthophosphate (ORP), (2) soluble reactive phosphorous (SRP), (3) total dissolved phosphorous (TDP) and unknown (UNK). At sites represented in the TADB and NADB databases, both phosphorous loading and wetland treatment performance varied. At the Iron Bridge, Florida, site, the mean influent and effluent dissolved phosphorus values (ORP) were 0.35 and 0.1 mg/L, respectively, while removal efficiency ranged from –16.3 percent to 73.6 percent. The long-term average total dissolved phosphorous removal efficiency based upon inlet and outlet concentration for the Houghton Lake, Michigan, system was 96.6 percent. In Listowel, Ontario, alum addition was part of the lagoon pretreatment process. The wetland treatment systems there also exhibited both negative (concentration increase) and positive (concentration reduction) soluble reactive phosphorous removal efficiencies ranging from 21.5 percent to 32.5 percent at Listowel 1 and 3, respectively.

FIGURE 4-25
Total phosphorus loading versus effluent phosphorus concentrations for the TADB FWS systems.

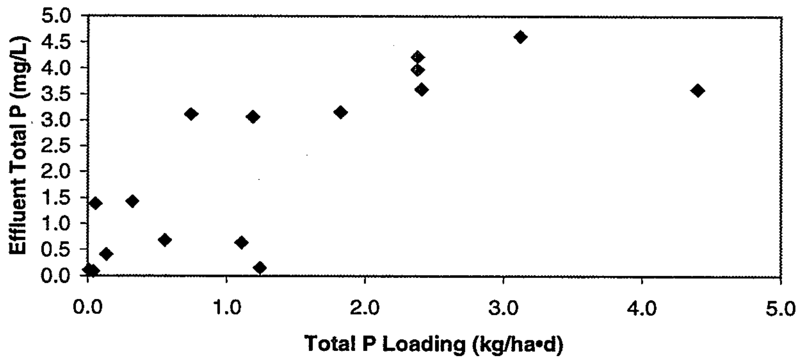
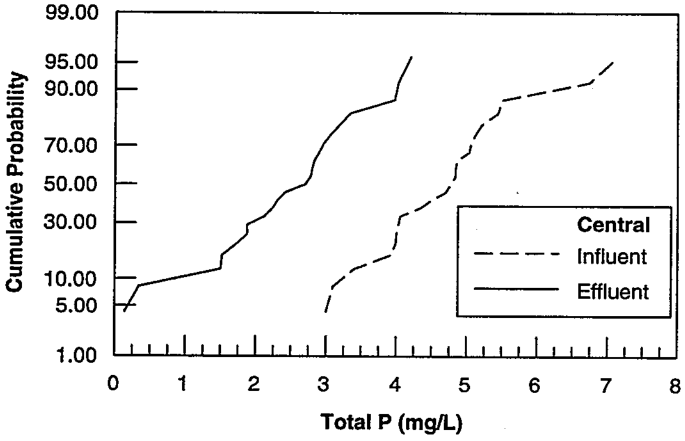


FIGURE 4-26
Cumulative probability distribution of monthly influent and effluent total phosphorus concentrations for Central Slough, South Carolina.

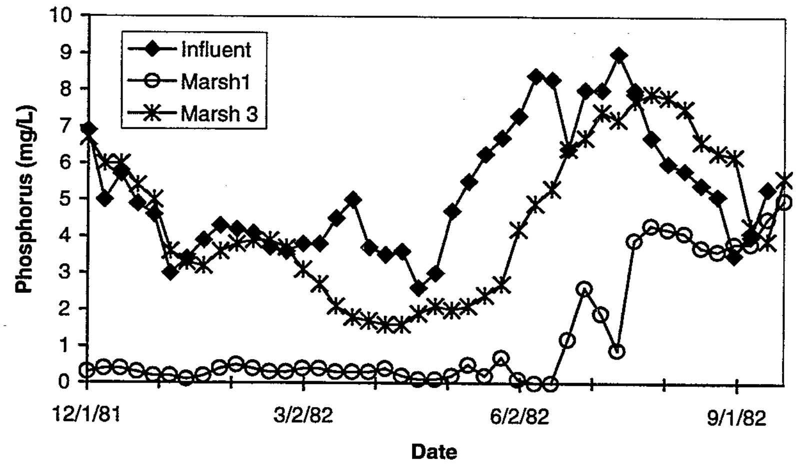


Temporal Phosphorus Performance

Phosphorus removal in FWS constructed wetlands follows a seasonal pattern in most temperate climate conditions. The form of phosphorus, type and density of the aquatic plants, phosphorus loading rate, and climate determine the amount of phosphorus removed in FWS constructed wetlands. Aquatic plants serve as seasonal reservoirs for phosphorus as they take up soluble reactive phosphorus (SRP) during the growing season, however, only a finite amount of SRP can be incorporated in the aquatic plants and plankton in the water column. In those temperate climates where senescence of aquatic plants occur in the fall, the majority of the biologically incorporated phosphorus is released back to the water column upon decomposition of the particulate organic phosphorus (POP) and detrital plant material.

Figure 4-27 shows an example of the pulsing of SRP for the conditions in Arcata, California. In this example SRP was loading at a rate of 0.15 kg/ha/day for a year (Marsh 3). A separate control cell, Marsh 1, was fed tap water (no phosphorus load, at the same HRT) at the beginning of the growing season (late January and early February). At a loading rate of 0.15 kg/ha·d, 1 to 2 mg/L of SRP was taken up by the aquatic plants and associated microbes through mid-summer. The stored phosphorus in the plant material was being released as the plants stopped growing and began to senesce, in late July. By early August, effluent SRP from Marsh 3 is 1-2 mg/L higher than the influent to the marsh cell. A cell received effluent for one year, with the same standing crop as Marsh 3, then received tap water for one year. This cell, Marsh 1, showed a significant contribution of SRP in the late summer as phosphorus was released from the plant material and the detrital layer.

FIGURE 4-27
Phosphorus pulsing, as illustrated in a pilot cell in Arcata, California. Marsh 1 received tap water until June 1982 (no phosphorus load), while Marsh 3 received oxidation pond effluent (Gearheart 1993).



Marsh cell 1 also shows that about 0.5 mg/L of SRP is always in solution even with no phosphorus inputs. The SRP is moving between various biological compartments, with relatively short half-lives, as different microbial communities dominate. The standing crop in this particular wetland was approximately 15,000 kg/ha·yr above-ground material.

Total Phosphorus Permit Compliance

Only five FWS wetlands had TP permit limits and associated data in the NADB. Permit limits for TP varied from 0.2 to 1.0 mg/L. The Reedy Creek, Florida, natural wetland systems had annual average TP limits in addition to monthly limits. Based on these limited data, it appears that FWS constructed wetlands can comply with very stringent TP effluent limits.

Four of the five systems in the NADB had 100 percent compliance with their TP effluent limits. The Iron Bridge, Florida, system met the most stringent limit, 0.2 mg/L, every month out of 63 recorded in the NADB with an average effluent TP concentration of 0.09 mg/L and a maximum of 0.16 mg/L during that period. The Orange County, Florida, hybrid wetland exceeded its monthly limit of 0.2 mg/L 5 months out of 37 months of record. The maximum TP value recorded during this period was 0.39 mg/L.

Fecal Coliform Performance

Database Assessment

As shown in Figure 4-28, there does not appear to be any general relationship between the influent and effluent concentrations of fecal coliform from the TADB systems. In general, the correlation between influent and effluent conditions was better for specific sites (Gersberg et al. 1989). For example, a consistent 2 to 3 log removal with a 6-day hydraulic residence time was measured in Cell 8 in the Arcata Pilot Project. The mean influent (from an oxidation pond) fecal coliform was 5,000 cfu/100 mL and the mean effluent concentration was 35 cfu/100 mL. The cumulative probability distribution for influent and effluent fecal coliform is shown in Figure 4-29. Fecal coliform removal was also found to be correlated with TSS removal in this system. In studies performed with MS-2 bacteriophage, virus removal appears to follow the removal of fecal coliforms (Ives 1988).

FIGURE 4-28
Influent FC versus effluent FC for the TADB systems.

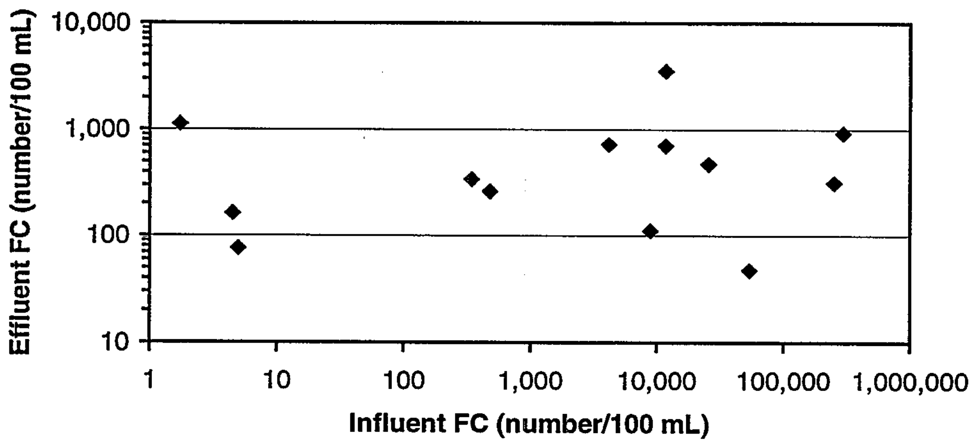
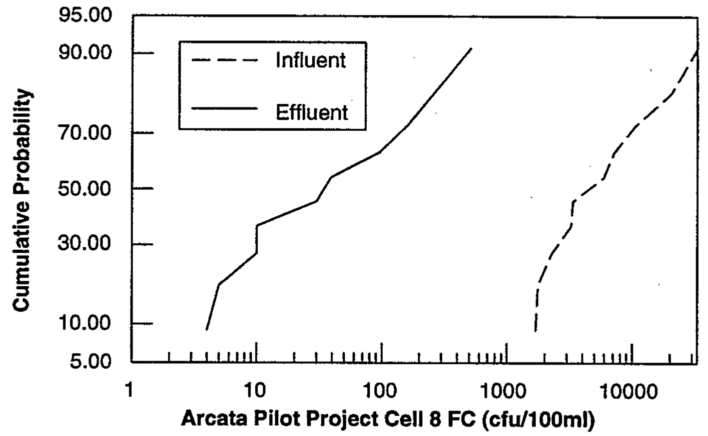
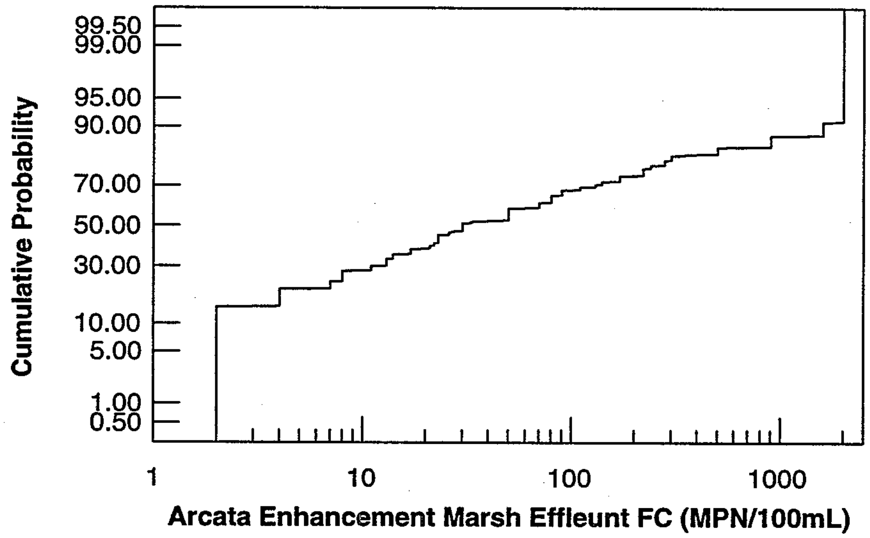


FIGURE 4-29
Cumulative probability distribution of influent and effluent fecal coliform from Arcata Pilot Project Cell 8, California (Gearheart et al. 1986).



Estimates of the internal production of background fecal coliforms in treatment wetlands is provided by those systems that receive disinfected influent. For example, the Arcata Enhancement Wetland receives chlorinated effluent, and during the period 1990-1997, the effluent FC was less than 500 MPN /100 mL about 80 percent of the time (Figure 4-30). A similar study on the same system during 1995-1996 showed that the effluent FC had a mean of 40 cfu/100 mL, was less than 300 cfu/100 mL more than 90 percent of the time, and that no sample exceeded 500 cfu/100 mL. While some of the differences between these two sampling results can be attributed to comparing MPN versus membrane filter results, they also indicate the variations that can occur over time at a single site.

FIGURE 4-30
Cumulative probability distribution fecal coliform from Arcata Enhancement Wetland, California (Gearheart 1998, unpublished data).



Temporal Fecal Coliform Performance

The considerable temporal variability in the effluent organism counts produced by treatment wetlands and conventional treatment technologies suggests the use of geometric averaging to determine monthly mean values from daily or weekly measurements. Even with geometric means, individual monthly values are frequently 10 times larger or smaller then the long-term mean for many treatment wetlands. As indicated by the preceding discussion, organisms in the wetland effluent did not necessarily originate with the incoming wastewater.

Fecal Coliform Permit Compliance

Only four FWS constructed wetlands had fecal coliform permit limits and associated data in the NADB. In each case, monthly effluent permit limits were 200 colony forming units (cfu)/100 mL; only one system met this limit 100 percent of the time (Apalachicola, Florida, with only 2 months of data). Percent compliance for the other four systems ranged from 22 to 83 percent. A maximum value of 27,000 cfu/100 mL was reported for one month from the Benton, Kentucky, constructed wetland, and maximum values of 2,600 to 5,800 cfu/100 mL were reported for Central, South Carolina, and Pembroke, Kentucky, respectively. Based on these limited data, it appears that most FWS constructed wetlands will have problems consistently meeting fecal coliform limits of 200 cfu/100 mL.

Metals

While some metals are required for plant and animal growth in trace quantities (barium, beryllium, boron, chromium, cobalt, copper, iodine, iron, magnesium, manganese, molybdenum, nickel, selenium, sulfur, and zinc), these same metals may be toxic at higher concentrations (Gersberg et al. 1984, Crites et al. 1995). Other metals have no known biological role, and may be toxic at even very low concentrations (e.g., arsenic, cadmium, lead, mercury, and silver).

Information from FWS treatment wetlands indicates that a fraction of the incoming metal load will be trapped and removed effectively through sequestration in plants and soils (Crites et al. 1995). A summary of published treatment wetland inlet/outlet metal concentrations from a variety of sites is presented in Table 4-4. For many metals, the limited data indicate that concentration reduction efficiency (EFF) and mass reduction efficiency (RED) correlate with inflow concentration and mass loading rate (Kadlec and Knight 1996). Wetland background metal concentrations and internal profiles are not well established.

TABLE 4-4
Metal removal data from free water surface treatment wetlands.

Metal	Wetland Type	Concentration (μg/L) In	Concentration (μg/L) Out	Mass Removal (kg/ha·yr)	Reference
Antimony	Constructed	0.45	0.20	0.6	Nolte & Associates 1998
Arsenic	Constructed	2.41	2.47	-0.1	Nolte & Associates 1998
Beryllium	Constructed	0.58	0.05	1.25	Nolte & Associates 1998
Cadmium	Constructed	43	0.6	2.4	Hendry et al. 1979
	Constructed	0.10	0.05	0.1	Nolte & Associates 1998
Chromium	Constructed	160	20	7.9	Hendry et al. 1979
	Constructed	3.4	1.5	4.5	Crites et al. 1995
	Constructed	1.57	1.13	1.0	Nolte & Associates 1998
Copper	Constructed	1,510	60	82	Hendry et al. 1979
	Constructed	8	3	11	Crites et al. 1995
	Constructed	7.87	3.48	10.4	Nolte & Associates 1998
	Natural	20.4	6.1	0.21	CH2M Hill 1992
Iron	Constructed	6,430	2,140	243	Hendry et al. 1979
	Constructed	205,000	6,300	29,900	Edwards 1993
	Natural	241	766	-4.3	CH2M Hill 1992
Lead	Constructed	1.7	0.4	3.1	Hendry et al. 1979
	Constructed	2.2	1.63	0.085	Edwards 1993
	Constructed	1.28	0.25	2.4	Nolte & Associates 1998
	Natural	2.0	5.5	-0.03	CH2M Hill 1992
Manganese	Constructed	210	120	5.1	Hendry et al. 1979
	Constructed	7,400	3,900	526	Edwards 1993
Mercury	Natural	<0.2	0.21	0.0001	CH2M Hill 1992
	Constructed	0.0112	0.0042	0.017	Nolte & Associates 1998
Nickel	Constructed	35	10	1.4	Hendry et al. 1979
	Constructed	7.5	3.8	0.8	Crites et al. 1995
	Constructed	6.26	7.10	-2.0	Nolte & Associates 1998
	Natural	17.0	9.1	0.14	CH2M Hill 1992
Selenium	Constructed	0.68	0.71	-0.07	Nolte & Associates 1998
Silver	Natural	0.36	0.53	-0.0005	CH2M Hill 1992
	Constructed	0.40	0.11	0.7	Nolte & Associates 1998
Zinc	Constructed	2,200	230	112	Hendry et al. 1979
	Constructed	36	11	60	Crites et al. 1995
	Constructed	36.85	6.71	71.3	Nolte & Associates 1998
	Natural	20.6	5.6	0.22	CH2M Hill 1992

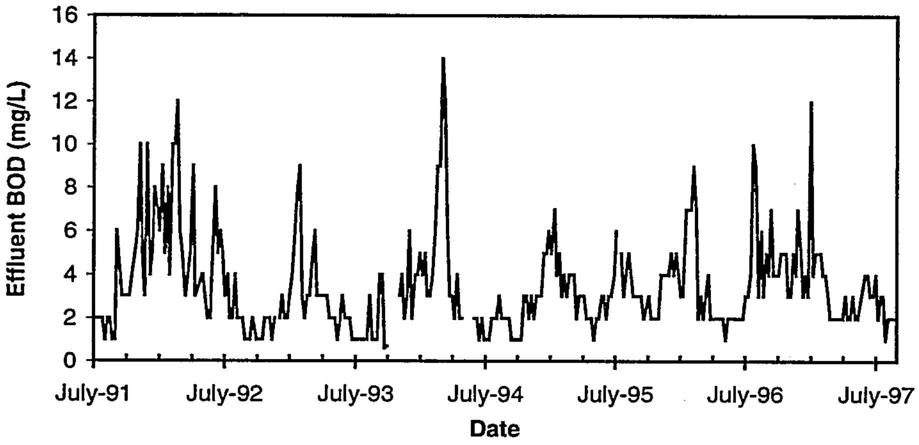
Other Performance Considerations

Wetland Background Concentrations

Wetland ecosystems typically include diverse autotrophic (primary producers such as plants) and heterotrophic (consumers such as microbes and animals) components. Most wetlands are more autotrophic than heterotrophic, resulting in a net surplus of fixed carbonaceous material that is buried as peat or is exported downstream to the next system (Mitsch and Gosselink 1993). This net production results in an internal release of particulate and dissolved biomass to the wetland water column, which is measured as non-zero levels of BOD, TSS, TN, and TP. Enriched wetland ecosystems are likely to produce higher background concentrations than oligotrophic wetlands because of the increased biogeochemical cycling that result from the addition of nutrients and organic carbon.

Background concentrations are not constant, but have a cycle of release that is a function of the biogeochemical cycle rates and external (other than wastewater inputs) factors. An example of this cycling can be seen in Figure 4-31 from the Arcata Enhancement Wetland. Six years of weekly BOD measurements show that for this system the background concentration varies between 1.3 and 4.0 mg/L. The higher values (3.5 to 4.0 mg/L) occur in the fall while the lower values occur in the summer. This variation is attributed to the accelerated decomposition of the vegetative material and to increased bird activity in the fall. Lower values in the summer are correlated with low decomposition rates (low recent litter production) and decreased bird activity.

FIGURE 4-31. Variation in effluent BOD at the Arcata Enhancement Marsh.



Treatment wetland background concentration ranges can be estimated from systems that are loaded at a low enough rate to result in an asymptotic concentration profile along a gradient of increasing distance from the inflow (several examples exist in the NADB). Long-term average annual outflow constituent concentrations for this selected group of FWS treatment wetlands are summarized in Table 4-5. Wetland systems typically have background concentrations within the ranges listed in Table 4-6.

TABLE 4-5
Long-term average annual outflow concentrations for lightly loaded FWS wetlands in the NADB.

	Concentrations, mg/L				
System	BOD_5	TSS	NH_4-N	TN	TP
Eastern Service Area, FL	1.2	3.0	0.07	1.45	0.09
Iron Bridge, FL	2.0	2.8	0.18	0.95	0.08
Bear Bay, SC	1.9	2.7	0.27	2.35	0.40
DesPlaines, IL	--	5.2	0.03	1.34	0.02
Hidden Lake, FL	3.0	13.0	0.05	0.66	0.16

Source: NADB 1993

TABLE 4-6
Expected range of background concentrations for constituents of interest.

Constituent	Unit	Concentration Range
5-day biochemical oxygen demand (BOD_5)	Mg/L	1 to 10
TSS	Mg/L	1 to 6
Organic N / TKN	Mg/L	1 to 3
Fecal coliforms (FC)	MPN/100 mL	50 to 500
TN	Mg/L	1 to 5
Ammonium N	Mg/L	less than 0.1
Nitrate N	Mg/L	less than 0.1
Total Phosphorus	Mg/L	less than 0.1

Natural Variability

Free water surface treatment wetlands demonstrate the same type of water quality variability typical of other complex biological treatment processes. While inlet concentration pulses are frequently dampened through the long hydraulic and solids residence times of a treatment wetland, there is always significant spatial and temporal variability in constituent concentrations. The stochastic character of energy inputs, rainfall, and the periodicity and seasonal fluctuation in ET contribute to the variable constituent concentrations often seen in treatment wetland effluents as can be seen in Figure 4-31, which shows the variability in effluent BOD concentrations over 7 years for the Arcata

Enhancement Marsh. Such variation can and should be accounted for by treatment wetland designers, operators, and regulators alike. If it is, FWS treatment wetlands can be utilized successfully and confidently in a communities overall wastewater management strategy.

SECTION 5

System Planning and Design Considerations

Planning Considerations

Like other wastewater treatment processes, FWS constructed wetlands perform within definable limits. These limits must be identified and summarized to allow designers to size FWS constructed wetlands that consistently achieve pollutant reductions from a known influent to a desired effluent concentration. Regression equations, areal loading rate methods, and simple first-order models are the most common tools used to summarize constructed wetland performance. With a general knowledge of performance expectations, the experienced designer can use these tools to specify characteristics such as wetland area, water depth, cell configuration, and plant selection to achieve desired treatment efficiency.

Consideration must also be given to specific constraints associated with the living, autotrophic ecosystems such as those that exist in FWS constructed wetlands. The natural processes that occur in FWS wetlands result in background concentrations for some constituents that may be higher than the influent concentrations of the same constituent. Knowledge of these background concentrations is important to avoid overly optimistic expectations for constructed wetlands performance. Additionally, a certain amount of statistical variability is inherent in wetland effluent concentrations, some of which is due to environmental factors (such as seasonal temperature and plant community changes) outside the control of the wetland designer and operator. Unless discharge permits are written to include this natural variability, the inevitable scatter associated with wetland effluent quality must be factored into design to avoid permit violations.

Some of the modeling tools and general considerations that are important to wetland planning, design, and sizing are described in this section. The models presented were developed with input output data collected from selected wetland treatment systems, which may, or may not be representative of the myriad of potential treatment wetland applications. Not unlike activated sludge or other conventional wastewater treatment process design, rate constants used in wetland models "lump" together the mechanisms and responses taking place to improve water quality because of the present constraints in data availability and quality control.

However, treatment wetland scientists, engineers, and practitioners are now in the process of refining existing relationships and exploring new sizing methods as new information is collected and made available. Expect models in the near future that consider the non-idealities of FWS wetland flow and/or utilize retarded rate constants to more accurately describe the principal removal/transformation mechanisms taking place within constructed wetlands. Once high quality data necessary to develop these relations are available, these models should provide more accurate insight into predicting the performance of FWS wetlands for a given source water, treatment volume, and/or treatment area. For now, it is paramount that individuals or entities wishing to design and implement FWS treatment wetlands for wastewater treatment utilize competent professionals experienced and abreast of the technology.

Role of Wetlands in the Watershed

The first step in assessing the feasibility of FWS constructed wetland is to identify the goals and objectives of the wetland within the watershed. Natural wetlands are an integral part of their watershed; functioning as water storage areas, nutrient sinks, and wildlife habitat. Once a minimum water quality is achieved, which protects public health and addresses ecosystem concerns, FWS can be used to provide considerable benefits beyond water quality improvement. These additional objectives should be integrated into the feasibility and planning process and ideally, incorporated into an overall master plan establishing restoration goals for the entire watershed and its receiving waters.

The process used to evaluate the feasibility of FWS constructed wetlands for water quality improvements and to function as landscape units on a watershed requires a sequence of assessments. The process is similar to the evaluation of conventional wastewater unit treatment processes because FWS constructed wetlands function similarly to them in terms of their ability to convert, remove, and store specific constituents. However, the process steps are dissimilar in that FWS constructed wetlands fulfill other functions and values as landscape units within a watershed. The procedure described below (Steps 1 through 12) incorporates evaluation of the possible additional functions of FWS constructed wetlands. The type of information required at each step and its relationship to the decision process is depicted graphically in Figure 5-1.

<u>Step 1</u> - Identify the goals and objectives of the project. In this initial step, the role the wetland will play in maintaining, restoring, or enhancing the beneficial uses in the receiving system is established.

<u>Step 2</u> - Characterize the wastewater(s) entering the FWS constructed wetland. Each type of wastewater or non-point water source has its own unique physical, chemical, and biological characteristics. A thorough characterization of the constituents and their concentrations combined with identification of pathogen indicators or pathogens should be conducted. This step should also include a thorough literature review and may require laboratory and mesocosm testing.

<u>Step 3</u> - Determine the discharge requirements and limitations. The discharge constraints coupled with the constituent properties determined in Step 2 would dictate the required effectiveness of treatment.

<u>Step 4</u> - Determine the ability for wetland processes to reduce, retain, and transform constituents. Mesocosm and bench scale treatability studies might be required prior to proceeding to the next step. Wetland treatability studies

usually require more time than most biological treatment systems because of the time it takes to develop the aquatic macrophyte standing crop.

Step 5 - Identify the roles the wetland can fulfill in the watershed given the constituent concentrations and treatment goals imposed upon it. Certain wetland roles may not be appropriate due to factors such as loading variations, types of constituents, and site location. The function and value of wetlands such as ecological (habitat/production), hydrological, biogeochemical, and educational can be important in determining the economic costs or benefits of the system.

Step 6 - Evaluate the site characteristics and constraints. The planning and design of a system is site specific. Once the type of system and the treatment goals have been established, the soil, vegetation, and hydrologic conditions necessary to achieve these goals are identified. The inherent characteristics of the site should be evaluated and compared to these requirements to determine the need for modifications and additions.

Step 7 - Determine the FWS wetland area required to achieve the treatment objectives. For planning purposes, the methods described in this section can be used as a preliminary estimate of the area required to achieve the treatment objectives identified in Step 3.

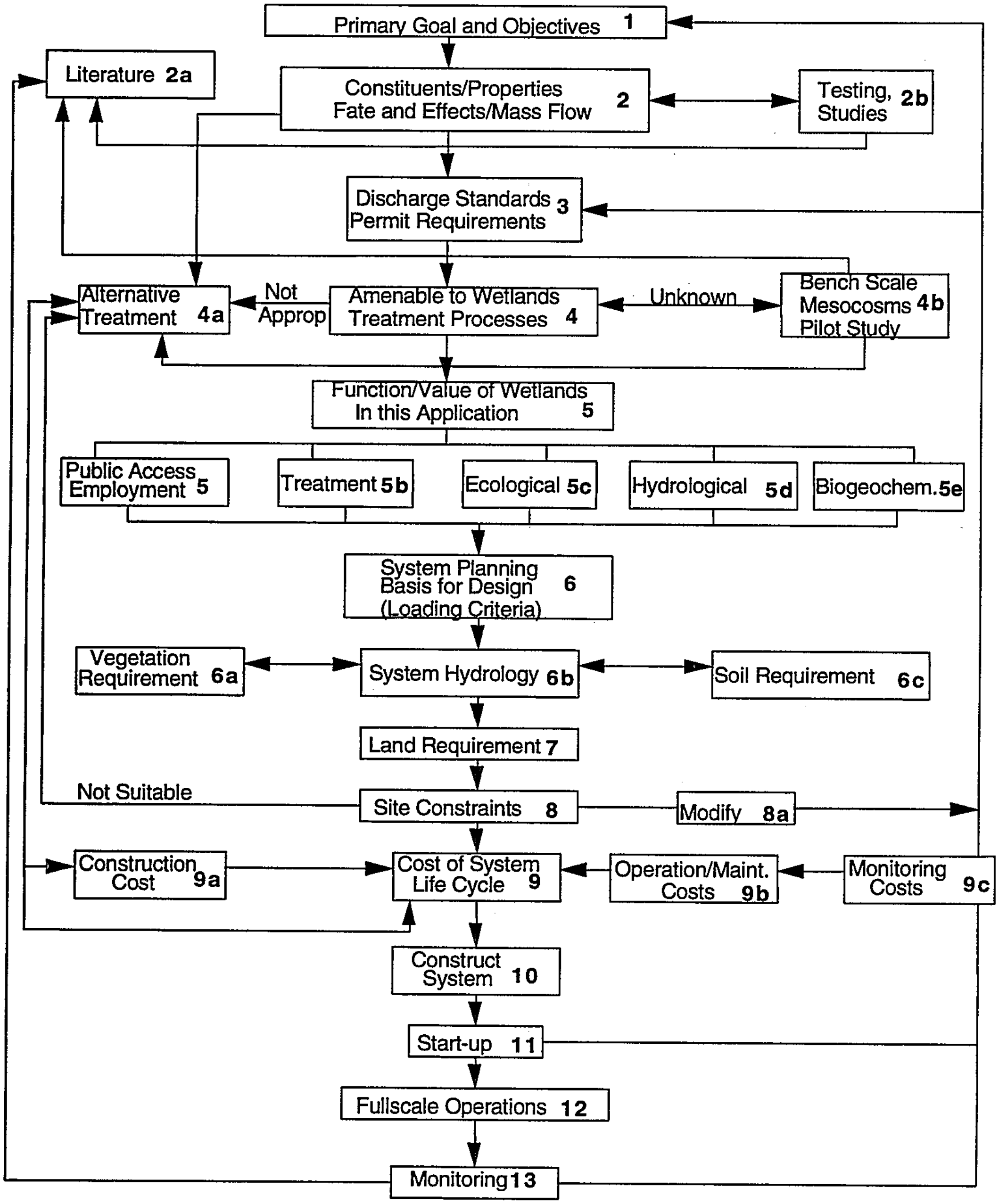
Step 8 - Evaluate alternate sites. The land capacity in terms of quantity and quality must be compared between alternate sites and technologies based upon constraints and capabilities.

Step 9 - Estimate the total cost of the system. The life cycle cost is a function of capital cost, and operational/maintenance cost distributed over a predetermined time base. The computed life cycle cost can be compared with alternative treatment systems or can be used to determine cost effectiveness and benefit/cost analyses. The value of the additional benefits, such as real estate, habitat, recreation, flood control, and water resource, should be included in the development of a total cost for the system.

Step 10 – Prepare construction and wetland system development plans. Wetland systems have several major differences from the construction of a conventional wastewater treatment plant. The primary difference is that aquatic macrophytes take time to develop into the requisite standing crop to support the treatment processes. Soils that support these plants are also critical to the start-up of the system. Preparing bid documents for the planting and maintenance of aquatic macrophytes should include the skills of a landscape architect and/or botanist with related experience. FWS constructed wetlands must also include flexible hydraulic controls for operational tasks such as isolating and draining cells. Inlet and outlet location and configuration should be considered at this step, as this can be critical to maximize treatment efficiency.

Step 11 - Plan the system start-up. The start-up of a wetland system might require changes in the hydrodynamics and density of vegetation. The start-up period for a FWS constructed wetland is regionally variable and can take from 18 to 36 months because it takes time for the plants to reach operational density.

FIGURE 5-1
Diagram of a methodology for determining the appropriateness of the use of a constructed wetland and the factors necessary for the design of a multi-use constructed free surface wetland.



Discharge permits for a wetland must reflect the lag time necessary to develop the requisite standing crop of vegetation to support treatment processes.

Step 12 – Full-scale operation requires determination of the placement and density of aquatic plants, inlet and outlet control structures, design hydroperiod, and design HRTs. The full-scale operation should have established background levels of soluble BOD, COD, ammonia nitrogen, etc. Full-scale operation could include determining procedures to store and/or drawdown the wetland system in anticipation of discharge constraints and/or peek monthly flow conditions. Procedures for control of vectors and nuisance mammals, vegetation management, etc., should also be developed and ready to implement.

Step 13 – Daily monitoring of influent flow and effluent flow should be conducted, and monthly average (weekly samples) BOD, TSS, coliform, and other (ammonia, nitrates, etc.) pollutant concentrations tracked. Vegetation coverage should be monitored annually along with the detrital accumulation (TSS, plant detritus, and floating litter layer). An inspection of the hydraulic integrity of berms, inlet and outlet works, and bottom (if required) should be performed annually. Under certain scenarios, monitoring for mosquito larvae and adults might be required during the mosquito-breeding season. The activities of other potential nuisance organisms such as nutria, beavers, and muskrats need to be monitored monthly as they can have a negative effect on effluent quality and wetland performance, in which case they might require management.

Additional Benefits/Habitat Considerations

Designers interested in providing habitat value in FWS treatment wetlands can turn to the ample literature on wildlife management to find clues to optimizing wildlife use. However, there is a significant amount of published and unpublished literature on habitat richness and wildlife populations in FWS treatment wetlands. Although these data have not yet been assembled and correlated to wetland design criteria, a treatment wetland habitat database is currently being prepared with funding from EPA's Environmental Technology Initiative to begin to fill this information void (Knight, in preparation, 1999). In the interim, a document published by EPA (USEPA 1988b) provides a general description of the habitat features of 17 treatment wetlands in the United States. USEPA has also published a book on Created and Natural Wetlands for Controlling Nonpoint Source Pollution that has chapters on habitat considerations (USEPA, 1993). Lastly, the habitat quality of two FWS constructed wetlands was evaluated by the EPA's Environmental Research Laboratory located in Corvallis, Oregon (McAllister, 1993).

Effluent Quality Considerations

Free water surface constructed wetlands produce a wide range of effluent qualities, depending on the influent characteristics, constituent operational loading rates, climate, and areal extent of the system. When designed and operated properly, FWS constructed wetlands perform within a predictable range of effluent values and meet their permit limitations. However, a limitation to using a FWS constructed wetland as a wastewater treatment system is the background concentration of constituents produced by external loading and internal wetland processes.

Natural background concentrations of BOD, COD, turbidity, total phosphorus, total nitrogen, and total and fecal coliform will control the effluent quality achievable using FWS constructed wetlands. The natural variation in the effluent from FWS wetlands is unique to each site and dependent upon the inlet/outlet configuration, hydroperiod, and seasonal factors controlling detrital decomposition, wildlife activity, and constituent influent loading.

The natural cycle of nutrients and the potential re-release of constituents incorporated in the wetland biomass must be considered in the effluent permit requirements for FWS constructed wetlands. In most cases, nutrient cycling and release follows seasonal patterns. The seasonal cycle of decomposition release or reduced microbiological conversion can be asynchronous with the critical water quality requirement for the receiving waters. For example, seasonal ammonia standards are often specified to protect receiving waters during periods of warm temperatures and low flow conditions. These receiving water conditions can occur during periods of high biological ammonia uptake in the wetland, resulting in the highest rates of ammonia removal and hence discharge limits can be attained.

In the case of coliform effluent standards, seasonal increases in coliform bacteria (total and fecal) may result from high bird populations in the wetland. If disinfection is required, the potential increase in wildlife populations in and near FWS constructed wetlands needs to be taken into consideration and may require seasonal permit exceptions. The extent and placement of open water, prime habitat for migrating waterfowl, is an important factor in minimizing increased coliform counts in the effluent.

Wetland Treatment System Objectives

The required effluent quality from a FWS constructed wetland is specified, in most cases, by the state water quality control regulatory agency. Effluent limitations are based upon (1) receiving water beneficial uses and, to some extent, by the receiving waters hydraulic and biogeochemical assimilative capacity and, (2) by reuse and reclamation guidelines specified for various reuse options. While FWS constructed wetlands have been shown to be effective wastewater treatment units, they do have treatment limitations due to factors such as seasonal nutrient cycling, plant decomposition, and bird activity. These limitations must be considered in both the design and the permitting of these systems.

Another critical treatment objective consideration is the wetland effluent discharge point. Most FWS constructed wetlands discharge to surface waters, but a leaky FWS constructed wetland can be designed to serve as both a treatment and disposal system (Nahar et al. 1998). Infiltration wetlands are designed to combine the horizontal processes in the FWS constructed wetland with the vertical processes through the sediment and soil to meet water quality objectives for either groundwater infiltration or surface water discharge. Examples of infiltrating FWS constructed wetlands performance can be found in the Hillsboro, Oregon, data and the Orange County Water District, Florida, and Tres Rios, Arizona, wetland demonstration projects.

Permitting

A major constraint on the use of many natural marshes is the fact that they are often considered part of the receiving water by regulatory agencies (Reed et al. 1995). Consequently, wastewater discharged to a natural wetland has to meet discharge standards prior to application. In Arcata, this obstacle was avoided by taking advantage of the "enhancement" clause of California State law regarding water quality, in which wastewater application can be allowed if enhancement of the existing wetlands can be shown. The distinction between natural and constructed FWS wetlands is not always clear, and the barriers to using natural FWS wetlands for treatment may also be applied to constructed FWS wetlands.

Historically, the use of natural wetlands in wastewater management in the Southeast occurred because of convenience or the lack of other reasonable alternatives. Only in the past decade have wastewater management systems incorporated design elements to optimize the wastewater renovation capabilities of wetlands. The use of natural wetlands for wastewater management may not be appropriate in many cases. Most situations will require site-specific analyses to determine site feasibility and acceptability based on existing natural wetland type, size, condition and sensitivity.

In general terms, the use of natural wetlands should be avoided when:

- The wetlands being considered are pristine wetlands and representative of unique wetland types;
- Projected impact to the wetlands would result in changes that would threaten the viability of the system; and/or
- Conflicts with other uses could not be mitigated adequately such as adjacent land use activity, availability, and cost of land.

Most natural wetlands are designated as "Waters of the United States." Such wetlands are either adjacent to other Waters of the U.S., or upon use, degradation, or destruction could affect interstate or foreign commerce, and as such, are afforded protection under the programs of the Clean Water Act. In Addition, other wetland protection programs must be considered when evaluating the use of natural wetlands.

Under the Clean Water Act, the four programs that can directly or indirectly affect wetland wastewater management decisions are:

- Construction Grants (Section 201)
- Water Quality Standards (Section 303)

- National Pollutant Discharge Elimination System (NPDES) Permits (Section 402)
- Discharge of Dredge/Fill Permits (Section 404).

For each program area, there are currently existing specific program regulations, guidance and procedures. However the use of wetlands for wastewater management has not been addressed specifically by any program and clear guidelines do not exist. Minimum criteria relating to waters of the U.S. that can be applied to wetland effluent discharges require that:

- Water quality standards be maintained;
- A minimum of secondary treatment be attained prior to discharging from municipal treatment facilities to natural wetlands considered waters of the U.S.;
- An NPDES permit for each discharger or discharge point; and
- A 404 Permit for the discharge of dredge and fill material into jurisdictional wetlands.

Regulations for the U.S. Environmental Protection Agency's (USEPA) three major wastewater management programs (Water Quality Standards, NPDES Permits, and Construction Grants) are designed for facilities discharging to lakes, streams, rivers, estuaries, or other free-flowing surface waters. Wetlands are different from these aquatic systems due to their role as a transition between fully terrestrial and fully aquatic systems. As such, wetlands are often hydraulically slow moving systems, as opposed to the free-flowing nature of most streams and rivers. Additionally, the functions and use of wetlands cover a broader range of ecological, water quality, and hydrological values. Because the regulatory guidelines and programs developed under the Clean Water Act's wastewater management programs did not acknowledge or address specific wetland considerations, they are usually not directly applicable to wetland wastewater management systems.

Although wetlands that are Waters of the U.S. cannot be classified for "waste treatment," they can be used in wastewater management as long as established uses are protected. Many wetland functions and values, (e.g., storm buffering, and water storage), however, are not covered by existing use classifications. Additional qualitative or quantitative criteria addressing wetland characteristics (e.g., hydroperiod, water depth, and seasonal influences) may be necessary and appropriate to protect wetland uses. Entities that choose to build constructed treatment wetlands for helping to meet advanced treatment requirements (e.g., the Tres Rios Project for meeting "excursions" by the 91st Ave. plant) that are also designed to provide high value wetland habitat for wildlife and public use may find themselves facing CWA §404 issues if they locate their system in existing wetlands or waters of the U.S. On the other hand, if they seek formal recognition of the habitat values for potential eligibility and use as wetland mitigation areas, they may also create long-term responsibilities to maintain these areas. Opportunities do exist, especially in the arid West, for such projects to involve the use of pretreated effluents to help restore degraded or former wetland systems. Further guidance on this and other policy and permitting issues associated with constructed wetlands can be found in "Guiding Principles for Constructed Treatment Wetlands: Providing for Water Quality and Wildlife Habitat, prepared by the Interagency Workgroup on Constructed Wetlands" (this document is available on-line at www.epa.gov/owow/wetland/constructed/guide.html").

Section 402 of the Clean Water Act authorized EPA and delegated to the states, administration of the NPDES Permit Program. This program requires a permit to discharge pollutants from any point source into waters of the U.S. Therefore, the discharge to wetlands considered as Waters of the U.S., or from treatment wetlands into a Waters of the U.S. requires the issuance of an NPDES permit. Important elements of the permitting process include the application process, establishing effluent limits, permit conditions and requirements, permit issuance, and compliance monitoring.

Alternatives which accompany the application for the NPDES permit for wetland wastewater systems include the use of a tiered approach for information requests and monitoring requirements based primarily on wetland types and hydraulic loadings. The use of performance criteria as a permit requirement to monitor wetland and downstream water quality is also suggested.

An important step in establishing effluent limits is determining whether the stream segment (or in this case the wetland) to which a discharge is proposed is classified as effluent limited or water quality limited as defined by EPA (1985). A stream segment that is effluent limited requires best available technology or secondary treatment. A stream segment that is water quality limited requires greater than secondary treatment. The task of establishing effluent limits in water quality limited situations is not straightforward. The use of water quality models may not adequately predict wetland responses to wastewater discharges and the use of an on-site wetland assessment will likely be necessary. The qualitative results of an on-site assessment then need to be related to quantitative or qualitative effluent limits.

Public Access

The ancillary benefit of wetland and riparian habitat associated with free surface constructed wetlands has given some communities the opportunity to allow total or limited public access to the wetland treatment facility after sufficient water quality improvement has been achieved. These ancillary (or value-added) benefits have allowed some communities to extend the public and environmental services of the wetland to other uses. Ancillary benefits include but are not limited to passive recreation, environmental education, green belts, mitigation wetlands, etc. Various states have their own guidelines and regulations concerning public access to wastewater treatment facilities. The addition of a passive recreation and/or an environmental education facility has, in many cases greatly enhanced the local and regional visibility of the project. This visibility and usage has in most cases resulted in community support related to the wetland treatment concept as well as for the community environmental service efforts, i.e., watershed planning, stormwater management, riparian/wetland corridors, etc.

Requirements placed on public access vary considerably from site to site even within the same state. In California for example, Arcata allows 24 hour, 365-day access to the Arcata Marsh and Wildlife Sanctuary, while Hayward restricts public access to environmental education visits, by appointment only. Much of these differences are due to the demographic and geographic settings of the two sites. Hayward is a highly urbanized area with no direct community management. Arcata, on the other hand, is a mostly rural area where intensive volunteer involvement and management efforts exist.

Public access, which does not disturb wildlife, is generally considered a favorable component of a project. Careful planning and design of a system can minimize human disturbances while maximizing the habitat value.

Hydrological Considerations

FWS constructed wetlands have been utilized successfully in a wide range of hydrologic, climatic, and geographic settings, establishing the general utility of FWS constructed wetland systems over an array of locations and conditions. Although these systems are robust enough to operate under a variety of scenarios, consideration must be given to the effects of local conditions on the performance. When possible, these local condition effects can be mitigated by design constraints.

Precipitation and Evapotranspiration

As described in Section 3, precipitation and evapotranspiration affect the performance of FWS constructed wetlands by altering the concentration of constituents in the wetland and by changing the volume of water transported through the wetland. In areas of high rainfall or during periods of high rainfall, the precipitation accumulated in the wetland can dilute effluent concentrations and reduce the hydraulic residence time (HRT). High evapotranspiration rates act in the opposite manner, concentrating the water quality constituents and increasing the HRT.

In arid regions of the United States, monthly net loss due to evapotranspiration can be as much as 25-400 mm. At typical hydraulic loading rates of 50 to 80 mm/d, the loss of water can concentrate the dissolved constituents 10 to 25 percent. At the same time, the nominal hydraulic residence time would increase proportionally. The opposite effect is observed in the wetter regions of the United States.

In regions with long dry periods, dramatic increases in coliform bacteria, total suspended solids, ammonia, and turbidity can occur at the start of the wet season. These increases in water quality constituents are due to bird fecal material and other particulates washed off the plants and into the water column at the beginning of the rainy season.

Groundwater

Free water surface constructed wetlands are normally designed to be isolated from underlying aquifers. For site design, the elevation of the seasonal high groundwater table and direction of predominant flow should be determined to ascertain potential problems with interception or berm failure. In the case of unlined FWS constructed wetlands designed to discharge through infiltration, groundwater monitoring will be necessary to measure constituent concentrations and hydraulic effects of the discharge.

Ice and Snow

In areas of significant snow cover or thick ice formation, free board is made available in FWS constructed wetland design and operation to allow the ice cover to serve as insulation over the water column. Ice formation requires an increase of 300 to 500 mm in the operating depth to maintain the design water column depth. In some cases, better effluent quality is obtained in the colder months due to the lack of external factor effects (wind, wildlife, etc.) and seasonal low contributions from internal sources such as plant litter and solids decomposition.

Engineering Considerations

Pre-Treatment Requirements

FWS constructed wetlands have pre-treatment requirements similar to other biological wastewater treatment processes. In Europe and the United States, this minimum appears to be that equivalent to primary and/or septic tank effluent. Floatable solids and large settleable solids should be removed from the influent wastewater. Excessive levels of oil and grease should also be avoided. Specific constituents or constituent loadings that may upset biological processes should receive pre-treatment. The wastewater delivery system should be designed to distribute influent evenly across the wetland cross-section to maximize the treatment volume available to remove settleable and suspended solids.

Also important to a FWS constructed wetlands are the incoming metal concentrations. While a FWS constructed wetland can remove and immobilize many heavy metals, if the system is designed for habitat enhancement, the potential for metals accumulation in the biota exists. In cases of high metal concentrations in the wastewater, a source reduction program and an industrial waste pretreatment ordinance may be more appropriate than a multi-use FWS treatment wetland.

Soils, Slope, and Subsurface Geology

The principal soils considerations in siting and implementing a FWS constructed wetland are the infiltration capacity of the soil and its suitability for berm construction. In most cases, FWS constructed wetlands are required to meet stringent infiltration restrictions. Specifications for infiltration losses from wastewater ponds and wetlands range from $1x10^{-9}$ to 7×10^{-6} mm/s depending on the state regulations for construction and groundwater protection. Systems designed to incorporate infiltration as part of the treatment and discharge process of the plant are an exception. In these cases, the underlying soil must have infiltration rates compatible with the design rates of discharge. In both cases, the native soils may need amendment or restructuring.

An additional soil consideration for FWS constructed wetlands is its suitability to establish and grow wetland plants. Aquatic macrophytes generally reproduce asexually by tuber runners. Soils with high humic and sand components are easier for the tubers and runners to migrate through, resulting in rapid plant colonization and growth.

FWS constructed wetlands can be built on sites with a wide range of topographic relief. Construction costs are lower for flat sites as highly sloped sites require more grading and berm construction. With proper design, however, high slope sites can possibly reduce pumping costs by taking advantage of the existing hydraulic gradients.

Percolation and Use of Liners

If the native soil does not have sufficiently low infiltration rates, amendment with clay or soil binders can be used. Another option for minimizing infiltration is installation of a geosynthetic membrane beneath the system (Kays 1986). Both of these requirements can add significantly to the construction cost of a FWS constructed wetland.

Inlet/Outlet Types and Placement

The hydraulic response of a FWS is dependent on several factors: vegetation type, amount and location, geometry of the system (especially as it might relate to dominant wind directions and velocities), and the type and location of the inlet and outlet works.

The inlet works should ensure a uniform distribution of the influent normal to the direction of flow. This can be accomplished in several ways. One technique is a manifold which extends across the inlet zone with adjustable ports located every meter or so. Another technique is to have several large inlet weirs (controls that allow to shut off the flow) which discharges into a mixing volume, which extends across the entire entrance of the wetland. This area tends to be deeper to minimize emergent plants and to ensure an even distribution of the influent through the aquatic plants in the wetland.

The outlet works can also be of several types. The outlet works serve both as water level controls and as collection points for the effluent. Not much work has been done comparing the various types of weir structures and locations as they relate to effluent quality. Geometry of the wetland cell has a lot to do with the number and type of outlet weir structures. In general, weir structures are placed every 8 to 25 m along the effluent collection zone located at downstream point(s) in a FWS constructed wetland. Similar to the influent collection/distribution zone, some systems have effluent collection volumes that direct flow to a weir collection/control structure. This type of system can produce variable TSS and coliform as both algal population and wildlife are attracted to this deeper clear water volume. Best successes have been observed where the aquatic plant communities are more, or less, contiguous with the effluent zone/control structure. Extremely high weir overflow rates in this type of system suggest that increasing total weir length might assist in improving effluent quality.

Wildlife/Habitat Consideration

A FWS constructed wetland utilized for treating municipal wastewater can also function as wildlife habitat, and in some cases where water quality permits it, constructed wetlands are being designed with wildlife habitat creation as a secondary or primary goal. This approach is similar to the role in which oxidation ponds are used by waterfowl and wildlife. Constructed FWS wetlands can provide incidental support of wildlife, or it can be enhanced by considering factors that encourage and support a wide range of wildlife communities. In the case of FWS constructed wetlands, the amount of open-water area and the types of submergent and floating macrophytes are positive habitat factors. The proportion and location of open water areas can also affect wetland effluent water quality. Based on pilot project work performed and subsequently used to design enhancement wetlands in Arcata, California (1986), it was shown that having 25 to 70 percent of the water surface dominated by submergent and floating macrophytes allowed optimal water quality and habitat enhancement objectives to be achieved (EPA 199b).

Another important design consideration for wildlife habitat is the inclusion of islands with low-sloped sides. Waterfowl and shorebirds can use the islands for feeding, nesting, and rest areas. Slopes of 1:4 to 1:10 around the island will encourage shallow zoned aquatic plants, while allowing easy access for aquatic fowl. Islands have been used effectively in many wetlands to support resident and migrating bird populations.

Environmental Impact

The following planning level considerations for the possible use of FWS constructed wetlands are important in communicating advantages and disadvantages of these wastewater treatment systems to clients, community members, and regulatory officials.

Land Use

The first major consideration for the use of FWS constructed wetlands is the land requirement and issues associated with general plans and zoning restrictions. Depending on the size of the community and the land uses adjacent to it, these could represent constraints or time-consuming requirements. Several possible strategies could be employed to expedite these issues. One successful strategy is to highlight the major advantages of FWS constructed wetlands; the multiple land use activities that can be assigned to their footprint. Overlays of land use activities, such as parks, passive recreation, wetland habitat, environmental education, green belts, possible wetland mitigation, open space, and viewshed corridors increase the public value of FWS constructed treatment system. These beneficial impacts can assist in mitigating the cost of the land for wastewater treatment.

Insect Vectors

Potential problems with insect vectors, particularly mosquitoes, are another major concern. Wetlands are prime habitats for mosquitoes and black flies, and are habitat for most of their major predators. Proximity of the wetland to houses and areas of intense use can become a siting constraint. For the most part, mosquitoes do not fly more than 400 m from their breeding area. Certain species, however, under the influences of wind direction and speed can disperse mosquitoes and black flies much farther. Regardless of location, mosquitoes will be present at some time of the year in any FWS constructed wetland. Serious consideration should always be given to implementing integrated pest management to control mosquito populations. Integrated pest management requires measures such as maintenance of an ecosystem that attracts and sustains viable populations of natural adult mosquito predators (dragonflies and damselflies, bats, swallows, frogs), and larval predators (carnivorous/omnivorous fish, and aquatic macroinvertebrates). Consideration should also be given to management of mosquito larval populations through the use of mosquito-specific larvicides such as those derived from the bacterium *Bacillus thuringiensis* or from a strain of *Bacillus sphaericus*. Chemical adulticide plans should also be formulated in case of mosquito generated public heath threats.

Odors

A FWS constructed wetland will have a seasonal odor associated with the normal decomposition of plant material and incoming settled solids. These odors will be more or less concentrated around the wetland as a function of micrometeorological factors such as wind speed, humidity, and lapse rate close to the surface. The odors associated with a FWS constructed wetland are not the same type or magnitude as the odors associated with a conventional wastewater treatment plant. Hydrogen sulfide is the predominant odor mixed with gaseous by-products of actinomycetes. The large area over which the odor is released tends to keep the concentration low, easily diffused, and dispersed. Odors can also develop if the influent wastewater is not properly introduced into the wetland.

Wildlife and Ecological Attractive Nuisances

While one of the major potential objectives of a FWS constructed wetland is to provide habitat value, some concerns are often voiced about the potential for attracting endangered species. At present, there is mixed information related to this issue. There is no state or federal law that exempts constructed wetlands from Endangered Species Act issues. There are examples where wastewater discharges support the habitat for endangered or listed species (e.g., pupfish in China Lake, California). For the most part, it is considered a net gain if a FWS constructed wetland becomes habitat for an endangered species. Oxidation ponds function similarly in many arid regions.

All FWS constructed wetlands provide habitat whether intended or not. FWS constructed wetlands that incorporate habitat features by design can attract large numbers of wildlife. One major potential problem is attracting too large a population of migrating birds. If the wetland supports large bird populations and water quality conditions are conducive to pathogen survival, then potential disease problems can develop (vibrio, clostridium).

The disease potential is highlighted at several wetlands in the San Francisco Bay area. For example, Hayward Marsh is the only source of freshwater on the Bay perimeter and as such, attracts large bird populations. It has limited vegetation cover that results in large open areas for resting, watering, and feeding. This provides large numbers of birds with opportunity to share common food sources and to come into close contact, effectively transmitting disease throughout the population.

The potential for introduction and spread of disease in migratory bird populations can be minimized. This is achieved by using a diverse assemblage of aquatic and riparian habitats, and by having the flexibility to manipulate the hydroperiod and flow rate into and out of the wetland.

Another major problem associated with constructed wetlands is the intentional release of domestic aquatic fowl and other domestic animals such rabbits, cats, and dogs. In the case of domestic ducks, their interbreeding with wild aquatic fowl presents a major wildlife problem. Feral cats may also be a significant problem as they feed on birds at the wetland. The issue of domestic species management requires advance plans be developed and implemented before problems escalate.

FWS Wetlands and Bird Strike Issues

Because of the great potential FWS wetlands have for attracting wildlife, specifically avian species, there exists a potential for conflict between animals and aircraft. In most cases, if siting a FWS constructed wetland within 5 miles from an airport, the habitat features must be in compliance with criteria set forth in 14 CFR Part 139. In brief, guidelines exist which govern the placement of habitat features within 10,000 feet and 5 miles from an airport that require developing a plan addressing wildlife hazards.

Wetland Sizing

As FWS constructed wetlands became recognized as a viable wastewater treatment process, a need arose for FWS design models. These models aid engineers in the process of FWS wetland design and performance assessment (e.g. wetland area requirements and effluent quality predictions).

Approaches to Sizing

The current trend in wetland design modeling is the development of simple mass balance or input/output models. These simplified models do not explicitly account for the many complex reactions that occur in a wetland, either in the water column or at interfaces such as the water/sediment interface. Instead, all reactions are lumped into one, overall reaction rate that can be estimated from FWS wetland input/output data. At this stage of wetland model development, more complex and theoretical wetland models—in which the kinetics of known wetland processes are described explicitly—are not possible, due to limitations in the existing wetlands data.

To date, a number of wetland design methods have been proposed for predicting constituent removals in FWS wetlands. The methods include fundamentally equivalent design relationships and equations presented by Reed et al., (1995), Kadlec and Knight (1996), Crites and Tchobanoglous (1998). The design relationships and methods have been used to predict the reactions (degradation or generation) of BOD, TSS, TN, NH_4, NO_3, TP, and coliform. An estimate of the wetland surface area can also be made by rearranging the relationships to solve for wetland area given constituent removal goals.

Design relationships are summarized in Table 5-1 along with new relationships proposed by Gearheart et al. (1998). The reader should keep in mind that none of the relationships presented in Table 5-1 are developed in this Technology Assessment to the extent needed to design a successful FWS treatment wetland. The reader is further encouraged to seek additional design information/insight

TABLE 5-1
Equations used to compute the performance of FWS constructed wetlands

Formula	Type	Definition of Terms
Reed et al. (1995) $\frac{C_e}{C_0} = \exp(-k_{VT}t)$	Volumetric	α = Delaying constant, temperature-dependant
		C_D = background BOD concentration contributed by decaying plants (g/m^3)
		C_e = effluent concentration (g/m^3)
		C_0 = influent concentration (g/m^3)
Kadlec and Knight (1996) $\frac{(C_e - C^*)}{(C_0 - C^*)} = \exp\left(-\frac{k_{AT}}{q}\right)$	Areal	C_1 = BOD concentration due to solubization of TSS and residual total BOD (1 to 65 days)
		C^* = background concentration (g/m^3) curve-fitting parameter
		k_{AT} = temperature corrected first-order areal reaction rate constant (m/yr)
Retardation Model (discussed in Crites and Tchobanoglous, 1998, Gearheart, 1999 [in preparation]) $C_e = C_0 e^{\left[\frac{-k_{VT}}{(1+\alpha t)}\right]} + C_D$	Volumetric BOD only	k_{VT} = temperature-dependent first-order rate volumetric reaction rate constant (d^{-1})
		K_{V1} = volumetric based solids/particulate BOD removal rate
		K_{V2} = volumetric based dissolved BOD removal rate – temperature-dependent
		q = nominal hydraulic loading rate (m/yr)
Sequential Model (Gearheart, 1999 [in preparation]) $C_e = C_0 e^{-K_{V1}t} + C_1 e^{-K_{V2}t} + C_D$	Volumetric Two-rates BOD only	t = theoretical hydraulic detention time (d)

from other sources such as those provided in Tables 2-2 through 2-4, or from competent professionals currently practicing in the field.

Regression equations have also been used to summarize system performance for a variety of constituents and physical parameters. General loading relationships have been used to predict removals for TSS, BOD, nitrogen, phosphorus and coliform.

To utilize one of the FWS design relationships or methods, it will be necessary to estimate or assume various parameters. Generally, the influent concentration, the expected or desired effluent concentration, and the flow rates are known from project goals and/or previous work. However, the remaining parameters need to be estimated from pilot project studies or assumed from literature values.

Assessment of Predictive Equations

In most of the existing FWS wetland design relationships, it is assumed that the hydraulics of FWS wetlands can be approximated by a plug flow reactor (PFR) model and the reactions of constituents are described by first order reaction kinetics. The use of the PFR to approximate the wetland hydraulics appears to be generally accepted by the wetland modeling community. However, there is ongoing debate over the appropriate form of the first order reaction rate constant.

The general relationship, assuming steady-state plug flow hydraulics and first order constituent removal, is:

$$\frac{dC}{dt} = -k_{app}C \qquad (5\text{-}1)$$

where: C = pollutant concentration (m/L^3),

t = mean hydraulic detention time (t), and

k_{app} = apparent first-order rate constant (t^{-1}).

This differential equation has the exact solution:

$$C_t = C_0 \exp^{-k_{app}t} \qquad (5\text{-}2)$$

where: C_0 = initial pollutant concentration at $t = 0$ (m/L^3).

The apparent first order reaction rate constant (k_{app}) can be a function of temperature so values are generally reported at 20° C. The k_{app} value can be adjusted to the desired temperature using a modified form of the van't Hoff-Arrhenius relationship:

$$k_T = k_{20}\theta^{(T-20)} \qquad (5\text{-}3)$$

where: k_T = apparent first order reaction rate constant at T degrees C [t^{-1}],

k_{20} = apparent first order reaction rate constant at 20° C [t^{-1}],

θ = empirical temperature coefficient, and

T = temperature at which k_T is adjusted.

The FWS wetland predictive equations presented in Table 5-1 are derived from the general PFR model (Equations 5-1 to 5-3). However, each of the models uses different concepts and approaches in defining the general PFR parameters (i.e., k and t).

The Reed et al. (1995) and Crites and Tchobanoglous (1998) relationships incorporate the adjusted nominal hydraulic detention time (t) through the wetland, and an apparent first order volumetric reaction rate constant. To utilize these equations, the depth, porosity and average flow through the wetland is required. The background pollutant concentration (C*) is not directly incorporated into these equations, but is included as a boundary condition (implied lower limit on effluent concentration) of the model.

The relationship proposed by Kadlec and Knight (1996) is based on the nominal hydraulic loading rate (q) to the wetland, and a temperature dependent first-order areal reaction rate constant. For some constituents, such as BOD, Kadlec and Knight report that the areal reaction rate constant is not temperature dependent. In this model the depth, porosity and water losses and gains through the wetland are not required, but lumped into the first order areal reaction rate constant. The background pollutant concentration, C*, is directly incorporated into the model equation.

Areal Loading Rate Method

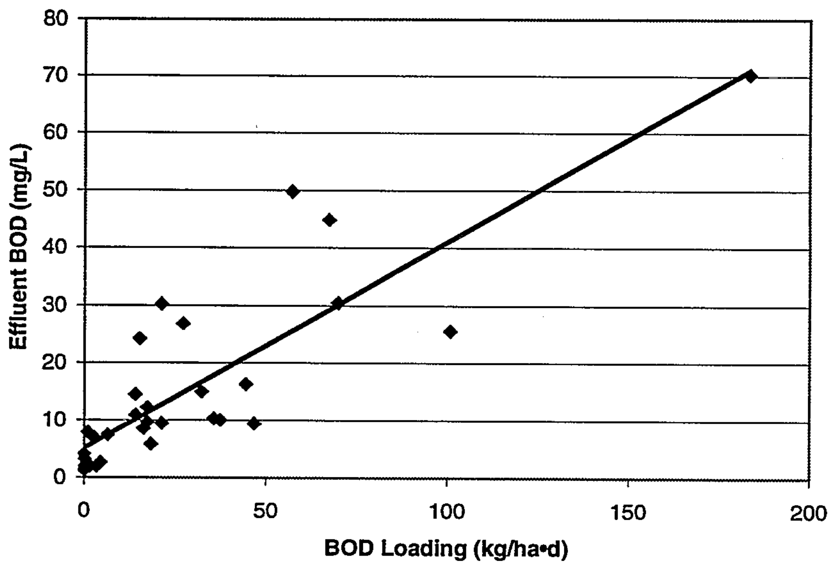
In the areal loading rate method, a maximum loading rate per unit area for a given constituent is specified. The use of loading rates is common in the design of oxidation ponds and can be used to give planning level surface area estimates for FWS constructed wetlands from projected pollutant mass loads. Areal loading rates are also used to check a FWS wetland designed using one of the above mentioned design models to ensure that the wetland is not overloaded. A range of typical influent concentrations, target effluent concentrations, and constituent areal loading rates for FWS wetlands are listed in Table 5-2. The suggested values given in Table 5-2 are based on data from the FWS wetland systems listed in Table 2-5. These rates can also be used to give a preliminary estimate of the FWS wetland surface area required for a given constituent loading, and to check wetland areas determined from equations in Table 5-1.

A typical areal loading design curve based on the long-term average performance of systems listed in Table 2-5 is shown in Figure 5-2. By knowing the areal loading rate, constituent effluent concentrations can be estimated from or compared to the long-term average performance data of full-scale operating systems.

TABLE 5-2
Range of areal loading rates for FWS constructed wetlands derived using data from FWS systems listed in Table 2-5 (Hydraulic Loading Rate for these systems ranged from 10 – 100 mm/day).

Constituent	Typical Influent Concentration (mg/L)	Target Effluent Concentration (mg/L)	Loading Rates (kg/ha·d)
BOD	5 – 60	5 – 20	10 - 50
TSS	5 – 60	5 - 20	10 - 60
TN	2 – 20	1 - 10	2 - 10
NH_4	2 – 20	1 - 10	2 - 10
NO_3	2 – 10	0.5 - 3	1 - 5
TP	1 – 10	0.5 - 3	1 - 5

FIGURE 5-2
Annual average areal BOD loading rate vs. annual average effluent BOD concentration for TADB systems.



Design Approach to Sizing

The approach to design of free water surface constructed wetlands should consider a wide range of local factors as well as general operational experience gathered on these systems. Design equations used to size FWS constructed wetlands summarized in Table 5-1 require the estimation of one or two parameters. Based upon observed data summarized in Chapters 3 and 4, "best-fit" parameter values vary greatly from site to site. The use of statistically derived, national parameters suffer from the inadequacies present in the data used to derive them as previously discussed. The equation parameters incorporate many factors and should be applied carefully when the setting and condition are different than those used to generate them. As discussed in Chapter 4, most of the systems in the database were underloaded and, therefore, are over-designed with respect to certain constituents in terms of areal requirement. None of the design formulas presented in Table 5-1 and used to determine wetland surface area requirements include the effect of inlet/outlet type and location, and vegetation type and distribution, which are potential determinants in wetland treatment effectiveness.

The first-order decay constant in all of the design equations is an apparent "k" value since it incorporates many factors including hydrological factors, temperature, solubilization factors, and removal/transformation processes. Over-designed wetlands mask the effect these factors have because the removal or transformation of a constituent is complete prior to the outlet. Since the performance of most of the wetlands in the database has been estimated from inlet and outlet samples this fact is reflected in the state of technology. As more experience is gained from multiple celled and/or systems in which samples are collected at intermediate points, a more useful database for the estimation of removal rate values can be developed. At present, the approach to design could include the use of the equations given in Table 5-1 with the resulting area checked against the empirical areal loading rates given in Table 5-2. This design approach is detailed in the EPA Wetland Design Manual (EPA, 1999).

Though plug flow is assumed for the purposes of FWS constructed wetland design, the actual wetland flow hydraulics will not follow an ideal model. The degree of deviation from plug flow of an existing FWS constructed wetland can be determined by tracer tests. One of the important results of a tracer test is the determination of the tracer detention time, defined as the centroid of the response curve. Tracer detention time is equal to the active water volume divided by the volumetric flow rate, and thus represents a direct measure of actual detention time. Comparison of the theoretical to the actual detention time is an important tool for evaluating the performance of existing FWS constructed wetlands.

The plug flow assumption is conservative in design if the degree of non-ideality (represented by the ratio of the theoretical plug flow to actual hydraulic detention time) in the designed system is less than that in the wetlands from which model parameters were determined (Hovorka, 1961; Kadlec and Knight, 1996). Because actual detention times are always less than the theoretical (plug flow) detention time, apparent removal rate constant estimates based on the plug flow assumption will be lower than the actual removal rate.

Using an apparent removal rate constant from one system for a different wetland with a different degree of actual to theoretical detention time can lead to serious over or under-design. For example, using tracer data developed at Treatment Marsh 1 (TM1) in Arcata, the observed hydraulic detention time was 84 hours. The theoretical detention time for this marsh is about 200 hours, nearly 250 percent longer. If an apparent removal rate constant computed based on the theoretical detention time is used for sizing a new system where the ratio of theoretical to actual hydraulic detention time is higher (say 3.5 :1), the new system will not meet performance expectations due to the relatively shorter actual detention time. The degree of non-ideality should be similar in wetlands with similar geometry, vegetation patterns, and hydraulic loadings. Note, the treatment wetland literature typically provides only apparent plug flow k values.

SECTION 6

Lessons Learned and Recommendations

A successful FWS treatment wetlands project requires that a number of other considerations be addressed which are just as important as the wetland process and design issues (e.g. water conveyance and wetland area) discussed earlier. Issues and considerations that are important in the implementation process for a FWS treatment wetland are described in this section. Items discussed include the need for more high quality wetland performance data and updating of wetland databases, potential nuisance conditions, open water/emergent vegetation areas, major components of wetland civil design and construction, issues surrounding wildlife enhancement wetlands, multiple benefits and public access, and general operation and maintenance considerations.

Information Management

New information from free water surface treatment wetlands is accumulating at a rapid pace. Between existing projects with ongoing monitoring programs and new projects that incorporate updated design features, the amount of useful information that could be applied to resolving technology issues is greater than can be accumulated and analyzed by individual wetland designers. Coordinated state or federal activities have not proven to be an effective method for keeping up with this accelerating information supply either. The most useful information has been generated by well documented moderately to highly loaded systems with cell-by-cell flow, depth, and constituent data.

Databases can provide a convenient method of accumulating and analyzing large amounts of treatment wetland design and operational data. Expansion, maintenance, and analysis of a FWS constructed wetland database is presented as a priority for future technology assessments. Research-level pilot studies provide the best method for testing the effectiveness of new treatment wetland design criteria. However, many pilot studies have failed to address new issues, and most have had such short operational periods that drawing general conclusions about the performance of a mature wetland from their data is difficult. New treatment wetland research efforts should consider focusing on some of the key technology issues that have been identified in this report.

Database Maintenance and Analysis

The initial NADB project began in 1991 and ended before completion in September 1993. This project captured a significant fraction of the wetland design and performance information available at that time, however; approximately 100 additional treatment wetlands in the U.S. and Canada were tentatively identified during that effort. It is now likely that up to 200 additional North American treatment wetland systems are not currently described in the database. In addition, the coverage and quality of data for those systems included in the NADB are suspect and can be incomplete in terms of using the data to evaluate system performance.

The depth of existing information displayed in portions of this technology assessment is testament to the potential value of an extensive design and operational performance database for wetland treatment systems. Intra-system data analysis allows determination of the effects of design variables on performance for major constituents of interest. Inter-system data comparisons allow the designer the opportunity to detect regional differences and differences due to variable water sources. The "data cloud" figures presented in this report reassure wetland practitioners that they can expect certain reasonable performance from treatment wetlands.

In spite of the limitations of the NADB, it can be used for a variety of purposes. One is to provide an inventory of how many treatment wetlands are "out there" and how they were built. This knowledge provides an understanding of how important this technology has become and to assess how rapidly it is growing, but does not require detailed operational data. A second goal, more in line with the purpose of a technology assessment, would be to assess accurately, wetland performance under a variety of design conditions. The NADB, as it is presently formulated and implemented, falls short of meeting this goal, in that insufficient information exists to optimize design of free water surface treatment wetlands. Since all of the presently available design models are approximations of system performance based upon the presently available limited data, the variability in empirical design relationships cannot be reduced until sufficient wetland data are available to document the effect of all design variables. More complex, multi-parameter design models can only be supported by analysis of detailed information from a number of long-term, research-oriented treatment wetlands. Lastly, ensuring that complete flow measurements are included for all systems is critical to the utility of the database in evaluating performance.

Additional funding should be sought for reformulating, updating, balancing, and editing the existing NADB. Most of the systems evaluated in the NADB are lightly loaded systems. Some of these systems have influent BOD and TSS values close to background values, resulting in periods of net negative pollutant removal. Efforts should be made to identify sites with higher loading rates to provide a more balanced view of the potential of the technology to treat wastewater. An initial effort could be completed over a 2-year period. The resulting updated NADB should be analyzed thoroughly and the results widely published. Practitioners in this field should be encouraged to maintain their own project data in an electronic form compatible with the reformulated NADB to allow rapid entry of new information.

Planning

Multiple Benefits and Public Access

The general public rather than individual landowners primarily receive benefits produced by wetland areas. After an appropriate level of pretreatment, wastewater introduced into a constructed wetland can sustain the wetland ecology and provide for multiple benefits, including public access (education, birdwatching, walking, jogging, and picnicking), two of the strongest endorsements for the use of this treatment process. The advantages of a multiple benefit investment in landuse can be a positive aspect of any FWS constructed wetlands project. These landuse types could include (1) parkland, (2) wildlife habitat, (3) environmental education, (4) open space, (5) greenways, (6) water reclamation storage, and (7) landuse set aside for future public use and treatment. These overlays of uses increase the societal value of the land investment made for treating wastewater. Public access is essential for communication and maintaining the multiple benefits of a FWS constructed wetlands project.

As a wastewater treatment system, FWS constructed wetlands have introduced a unique management opportunity. If the wetland system has multiple benefits, such as education, recreation and research, a public access policy needs to be developed specifying public use guidelines. Public access to a wastewater treatment facility is normally restricted due to the potential risk associated with pathogens present in the wastewater. Many states have specific regulatory constraints concerning public access to wastewater treatment facilities.

Clearly, public contact with raw or untreated wastewater is a potential human health threat that must be eliminated from both conventional and wetland treatment systems. If appropriate pre-treatment is provided a responsible public-use / access policy can be developed which allows for many of the potential ancillary benefits of a wetland treatment system to be realized. In assessing the upstream treatment processes with respect to potential health risks in wetland systems placed further downstream in the treatment train, it is important to take into consideration that there is a distinction between secondary treatment processes with respect to pathogen removal. For example, there is a measurable difference in the potential public health risk to public access between a lagoon secondary process with 20 to 40 days of HRT and an activated sludge system where HRT's are typically 0.3 to 0.5 days. This scenario results in ratios of 60:1 versus 120:1 in the difference in exposure to natural disinfection processes (Gearheart, Personal Communication).

The level of pretreatment necessary for public contact may be achieved at the end of the conventional process or at some point within the treatment wetland complex. In the latter scenario, restricting public access may be more challenging but is not impossible. In either case, the goal to eliminate the likelihood of human pathogen transmission to those visiting the facility should be paramount.

Some communities have successfully convinced regulatory agencies to allow full and/or limited public access to the wetland component of the wastewater treatment facility after adequate pretreatment has been attained. Public access is provided for or encouraged at a number of treatment wetland sites in the communities of Arcata, Hayward, and Martinez, California; Cannon Beach, Oregon; Incline Village, Nevada; Phoenix and Tucson, Arizona; and Iron Bridge and Everglades National Park, Florida. Limited published data concerning public use of these sites are available, including a thesis (Benjamin 1993) in which it was reported that there are about 90,000 visitors per year over the 2,000-acre Hayward Marsh, and 140,000 visitors per year at the Arcata Marsh and Wildlife Sanctuary. In 1998 the visitors to the Arcata, California, system had increased to approximately 180,000 (Gearheart, Personal Communication).

Examples of projects with significant wetland habitat values and wildlife usage are featured in *Constructed Wetlands for Wastewater Treatment and Wildlife Habitat – 17 Case Studies* [EPA 832-R-93-005, 1993.] Available information on such benefits is summarized in Treatment Wetland Habitat and Wildlife Use Assessment: Executive Summary [EPA 832-S-99-001; June 1999]

Environmental Education and Interpretation Centers

Wetland treatment systems present an excellent focus and facility for implementing community wide environmental education dealing with water conservation, pollution prevention, wastewater treatment, water reclamation, wetland ecology, watershed management, and energy conservation. The wetland site can be designed to incorporate public access (limited or full), esthetically pleasing viewsheds, riparian and upland fringe areas, and physical structures for interpretative purposes. All of these components can complement the wastewater treatment objectives of a city and increase the public stewardship of water resources through awareness, protection, and participation in their natural surroundings. One of the strongest cases for incorporation of these benefits can be seen in subsequent support for water quality and watershed protection requirements.

Some communities have constructed Interpretive Centers, which are the focus of much of the organized environmental education occurring at FWS constructed wetlands. Examples of interpretive centers can be found at Hayward Marsh, California (East Bay Park District), and the Arcata Marsh and Wildlife Sanctuary, California (Friends of the Arcata Marsh). Many other wetland systems have incorporated informational signs into trail system(s) surrounding the wetlands for environmental education. Local educational institutions can use FWS constructed wetlands as a field trip site for biology, wildlife, and engineering classes. In some communities, the wastewater utility forms partnerships with school districts to allow use of the wetland and interpretative center for environmental education. This component of a FWS constructed wetland allows for unique and creative sharing of resources and spaces to meet larger community needs.

Open Water/Emergent Vegetation Ratio

Providing adequate open water areas (open water/emergent vegetation ratio) is an important, but often overlooked, component in the design and implementation of FWS constructed wetlands. Historically, many FWS constructed wetlands were designed and built as fully vegetated basins with no open water areas. Some systems configured in this manner experienced problems with very low water column dissolved oxygen levels, incomplete nitrification, odor production, and vectors, primarily mosquitoes.

Many natural wetlands contain a mix of open water and emergent vegetation areas and they are as important for water quality reasons as they are for wildlife purposes. Open water areas provide many functions such as reoxygenation of the water column from atmospheric reaeration and algal photosynthesis, and habitat and feeding areas for waterfowl, as well as allowing for the predation of mosquito larvae by fish and other animals. Open water areas in FWS constructed wetlands also reduce BOD concentrations and improve nitrification of ammonia in wastewater because of the increased oxygen levels. In most cases, it is recommended that a FWS constructed wetland incorporate a mix of shallow vegetated and deep open-water areas that should result in a more complex, dynamic, and self-sustaining wetland ecosystem that more closely mimics a natural wetland.

The ratio of open water to emergent vegetation depends on the function and goals of the FWS constructed wetland project. For constructed wetlands whose primary function is wastewater treatment, the location and amount of open water is a function of the nitrification requirement for that system. Open water (submergent and floating aquatic plants) supports nitrification processes while minimizing the internal carbon load. If land area is at a minimum, and/or costs are to be kept low, then a minimal amount of open water area should be provided. However, if land availability is not an issue, then a maximum amount of open water area can be provided. Recommended open water to emergent vegetation requirements range from 0 to 30 percent for treatment wetlands and 40 percent or greater for enhancement wetlands.

While higher open water may be desirable, treatment wetlands can operate successfully at the suggested lower limits if constrained by land availability and/or construction costs. Generally, enhancement wetlands will be designed with larger open water areas for waterfowl and other wildlife than treatment wetlands with water quality improvement as its only performance criterion.

Two methods can be used for creating open water areas: (1) excavate zones that are deep enough to prevent vegetation growth, and (2) periodically raise water levels to a depth that limits vegetation growth. Thus, wetland design and operation can be used to control the types of plant communities that exist in FWS treatment wetlands. The type of macrophytes (i.e. emergent, submergent, and floating) can be controlled to some extent by the design operating water depth. Water column depths of 1 to 1.5 m planted with submergents, such as *Potamogeton spp.*, will not be encroached upon by emergent macrophytes like *Scirpus spp.* and *Typha spp.* If the water column depth is between 0.2 to 0.6 m and planted with emergent vegetation, such as like *Scirpus spp.* and *Typha spp.*, they will prevail over submergents and fill in the surface area through rhizome and tuber propagation. Alteration of water depth is a determining factor in establishing various aquatic macrophyte communities to meet both water quality and habitat objectives. A list of common wetland vegetation species and typical growing depths was provided in Section 3.

Large open water zones that are not shaded by emergent or floating macrophytes can allow significant blooms of phytoplanktonic or filamentous algae to establish in FWS wetlands. However, if the open water areas are designed for less than 3 to 4 days open water travel time, then algal growth should not occur, as the growth cycle of algae is approximately 7 days. Finally, if open water zones are located immediately adjacent to the outlet, the wetland may not be able to consistently meet stringent standards for BOD, TSS, or nutrients due to the export of algal solids. For this reason, it is recommended that a large vegetated zone exist (emergent or floating aquatic plants) at the outlet of a FWS constructed wetland to reduce sunlight penetration of the water column.

Site Topography and Soils

Pre-existing topographic, geological, and soil chemistry conditions can greatly affect wetland cost and performance. Excessive site relief creates large earthwork volumes for a given wetland area, significantly increasing construction costs. Surface and subsurface geologic conditions can also increase costs by requiring removal of rock or by presenting the need for liner materials to reduce groundwater exchanges. For the most part, level land with clay soils affords the best physical setting for a FWS constructed wetland. However, potential wetland sites with other conditions can be used, but may require more complex engineering, earthwork, and construction techniques, and the use of geotextile membranes.

Another consideration in the construction of a FWS constructed wetland is the soil required to support the emergent aquatic plants. Substrate for this vegetation should be agronomic in nature (e.g. topsoil), well loosened, and at least 150 mm deep. If this type of soil exists at the site it can be scraped off prior to excavation and saved, otherwise it can be imported from offsite. After the wetland basin, berms, and other earthen structures are constructed and the liner is installed (if required), the agronomic type soil can be placed back into the excavated region. This pre-conditioned substrate will greatly increase the rate of plant growth and extent of plant community coverage.

Another concern regarding soils is elevated concentrations of organic carbon, organic nitrogen, or phosphorus, which may result in increasing concentrations (negative removal efficiencies) between the wetland inlet and outlet following system startup. This potential problem can be anticipated during design and managed effectively by initial batch flooding to allow desorption and refixation (SFWMD unpublished).

Wetland Hydraulics

Inlet/Outlet Structures

Placement and type of inlet and outlet control structures are a critical feature in FWS constructed wetlands. Within the general loading guidelines, control structures are the most important feature after shape, in terms of wetland treatment effectiveness and reliability. To minimize short-circuiting in a FWS constructed wetland, two guidelines concerning inlet/outlet structures are critical: (1) effective distribution of inflow across the entire width of the wetland inlet and (2) uniform collection of effluent across the total wetland outlet width. These guidelines will also minimize localized velocities around inlet/outlet structures, thus reducing potential resuspension of settled solids. It is important that any outlet structure be designed so that the wetland can be drained completely, if required. Listed below are some of the common types of wetland inlet/outlet systems in use today, and general guidelines regarding their design.

Two types of inlet/outlet structures are commonly used in FWS constructed wetlands. For small or narrow wetlands perforated PVC pipe can be used for both inlet and outlet structures. The length of pipe should be approximately equal to the wetland width, with uniform perforations (orifices) drilled along the pipe. The size of the pipes, and size and spacing of the orifices will depend on the wastewater flowrate and the hydraulics of the inlet/outlet structures. It is important that the orifices be large enough to prevent clogging with solids, but small enough to provide uniform distribution along the length of the pipe. In some cases, the perforated section(s) of this type of inlet/outlet structure can be covered with gravel to provide more uniform distribution or collection of flows. Where the local climate permits, the use of an exposed, accessible inlet and/or outlet manifold is recommended for FWS wetlands to facilitate maintenance, except in the cases where public exposure is an issue.

For larger wetland systems, multiple weirs or drop boxes are generally used for inlet and outlet structures. Weirs or drop boxes are usually constructed of concrete. These structures should be located no greater than every 15 m apart across the wetland inlet width, with a preferred spacing of 5 to 10 m apart. The same spacing requirements apply for the outlet weirs or drop boxes. Depending on the source of the wastewater influent, the inlet weirs or drop boxes can be connected by a common manifold pipe, or directly to the wastewater influent source (a common arrangement for wetlands adjacent to oxidation ponds). Whatever the configuration, it is important that the hydraulics of the manifold and weirs be analyzed to ensure that uniform distribution occurs. Simple weir or drop box type inlet structures are relatively easy to operate and maintain, but generally provide less potential for solids settling in the inlet zone than a perforated pipe inlet with its axially distributed load.

Depending on the type of wastewater influent, the inlet structure outflow point can be located below or above the wetland water surface. Oxidation pond effluent, for example, which is high in algal suspended solids, should be introduced near the surface to allow for maximum settling, autoflocculation, and predation to occur. Primary or secondary treated effluents should be introduced below the surface if flocculated solids are expected, or if oil and grease, and/or primary solids are expected.

Outlet structures represent an operational control feature that can affect wetland effluent water quality. It is important that outlet structures have a wide range of operating depths. By adjusting the outlet structure, both the water depth and hydraulic detention time can be increased or decreased. The quality of wetland effluent found in the upper layers of the water column is generally of higher quality then water from lower depths, especially in terms of dissolved oxygen, TSS, BOD, and hydrogen ion (pH). However, the differences in water quality between water depths can be highly variable, and in some instances water from lower depths can be of higher quality than upper layers. An outlet structure design that allows for maximum flexibility of collection depths is recommended. With this type of design, the outlet structure can be raised or lowered to draw wetland effluent from the water depth with the best water quality.

Flow Measuring Devices

After analyzing the NADB, it became apparent that many existing wetland systems do not have flow-measuring devices. Even if accurate estimates of inflows and/or outflows to the treatment plant are known, internal flow distribution to individual wetland cells was not known or measured. Without accurate flow measurements to individual wetland cells, it is impossible to determine actual flowrates and hydraulic detention times to each cell, thus making flow adjustments difficult. It is recommended that some type of flow measuring device be installed in all FWS constructed wetland projects and further, separate flow measuring devices should be provided at each inlet and outlet for multiple wetland cell configurations. Typical examples of flow measuring devices include simple V-notch or rectangular weirs, and more sophisticated Parshall flumes. Depending on the size and layout of the wetland, flow measurement devices can and should be incorporated directly into inlet/outlet structures.

Internal Drainage

In the event a FWS constructed wetland needs to be drained, the wetland bottom should have a minimum slope of 0.1 percent to assist in drainage. Drainage may be required for maintenance reasons such as liner repair, vegetation management, and berm repair. Deeper channels may be required to allow for drainage and/or continued use when serial cells are taken out of service. Channels can also be used to connect deep-water pools, which may have been designed into the project to afford open water for waterfowl, or to provide refuge for fish and aquatic invertebrates during drawdown for maintenance.

Engineering

FWS treatment wetland construction has several planning issues based upon soil type, slope of the land, and cell configuration and shape. Other issues are associated with the civil engineering aspect of the design, such as impermeable barriers and liner materials, berm construction and specifications, inlet/outlet structures, flow measuring devices, internal drainage, sediment settling zone, and wetland planting. Many of these issues should be considered during the site selection process, as they may become difficult or more costly to correct later in the actual design and construction of the FWS constructed wetland. For the most part, the construction/civil engineering requirements are similar to other earthen water quality management systems such as sedimentation ponds, oxidation ponds, and sludge lagoons. The more important construction/civil engineering design issues that need to be considered in a FWS constructed wetlands project are as follows.

Berm Construction and Specifications

The height and width of berms or levees around FWS treatment wetlands is important for a number of reasons. First, the berms must be able to contain all design flows over a range of roughness conditions, including significant headloss through densely vegetated wetland cells with high aspect ratios. Secondly, the berms must be high enough to account for normal or excessive rates of solids deposition and peat building over the planned life of the wetland. The third consideration is the need to hold and release peak wastewater inflows, especially from collection systems with high infiltration and inflow rates or to planning storage for systems that do not discharge during periods of the year (typically winter). A fourth consideration is the need to protect berms from damage by animals and root penetration.

Berms containing FWS wetland cells are generally built with 3:1 side slopes, unless the soil characteristics allow for a steeper slope configuration, and a minimum of 0.6 m of freeboard above the average operating water depth. For wetlands that will receive high peak inflows, additional freeboard may be required to ensure that berm overtopping does not occur. All external berms should have a minimum top width of 3.0 m that should provide an adequate road wide enough for most standard service vehicles to operate on. In some cases, internal berms can have smaller top widths, as routine operation and maintenance can be carried out by small motorized-vehicles, such as ATVs. Road surfaces should be of the all weather type, preferably gravel to minimize direct runoff into the wetland.

Berm integrity is critical to the long-term operational effectiveness of FWS constructed wetlands. Common berm failure mechanisms include burrowing by mammals such as beaver and muskrat, and holes from root penetration by trees and other vegetation growing on or near the berms. Several design features can eliminate and/or minimize these problems. The insertion of a thin impermeable wall, or internal layer of gravel, installed during construction, can minimize mammal burrowing and/or root penetration. Also, planting the berm using vegetation with a shallow root system can be effective. Unlike oxidation ponds, berm erosion in FWS constructed wetlands from wave action is generally not a concern due to the dampening effect of the wetland vegetation.

In the design and site selection process, an important consideration is the amount of additional area required for berms. In general, the higher the length to width ratio for a FWS constructed wetland, the more area will be required for the berms and for the entire wetland system. This increase in required total wetland area to accommodate berms is more pronounced for smaller wetlands (less than approximately 10 ha) than for larger wetlands, but in both cases manifests itself as increased construction costs.

Wetland Configuration and Shape

There is substantial evidence, in both the design of oxidation ponds and FWS constructed wetlands, that a number of cells in series can consistently produce a higher quality effluent. This is based upon the hydrodynamic characteristics of "tanks in series," where constituent mass is gathered at the outlet end of one cell, and redistributed to the inlet end of the next cell. This process also minimizes the short-circuiting effect of any one unit, and maximizes the contact area in the subsequent cell. In addition, multiple treatment wetland cell configurations operated in parallel add operational flexibility to the overall treatment process and can facilitate maintenance activities. For treatment and water quality purposes, a FWS constructed wetland system could consist of a minimum of 2 to 3 cells in series with the capability of taking one cell out of service; however, the effects of headloss and inlet/outlet structures must be considered for systems constructed in this manner.

The shape of a FWS constructed wetland can be highly variable depending on site topography, land configuration, and surrounding landuse activities. FWS constructed wetlands have been configured in a number of shapes, including rectangles, polygons, ovals, kidney shapes, and crescent shapes. There is no general data that supports one FWS constructed wetland shape as being superior in terms of constituent removal and effluent quality, over another shape. However, design issues such as hydraulic detention time, short-circuiting, headloss, inlet/outlet structures, internal configurations, etc., do significantly affect wetland effluent quality, and some wetland shapes could potentially compound these problems over others.

Sediment Storage Zone at Inlet

Incoming settleable total suspended solids loadings are often waste stream specific and are removed by discrete settling in the inlet region of a FWS constructed wetland. Because a significant portion of the solids can often be removed in the inlet area of the wetland, every effort should be made to optimize the treatment potential of this region. It is recommended that some type of open water (settling zone) or solids retention area is provided in the inlet region of a FWS constructed wetland.

A settling zone could consist of an open water area that exists across the entire width of the wetland inlet. A possible guideline is to design a settling zone such that it provides approximately 1 to 2 days hydraulic detention time at the average wastewater flowrate, as most of the suspended solids are removed in the first 1 to 2 days of detention time in a FWS constructed wetland (refer to Section 4). The settling zone should be deep enough to provide adequate accumulation and storage of settled solids, and to prevent the growth of emergent vegetation, such as bulrush and cattails. The accumulated solids will slowly decay and reduce in volume over time. However, at some time in the future the accumulated solids may need to be removed from the settling zone.

Finally, inlet structure location and design will directly influence inlet velocities in the settling zone. Velocities in the outlet zone are functions of the cell geometry, vegetation pattern, and inlet/outlet type and location.

Wetland Planting

One of the most important considerations in the construction of a FWS constructed wetland is the lead-time necessary to develop a fully vegetated wetland. This factor enters into effluent compliance schedules and start-up periods. The planting strategy can determine the length of time it will take to reach functional densities of wetland vegetation. In general, the greater the initial planting density, the sooner the vegetation stands are developed. However, greater planting densities can also lead to greater planting costs. The source and type of planting material is also a major concern. Wetland planting success is highly dependent on the skills of the planting contractor, the type and quality of planting material, the soil matrix, and the time of planting. At best, it can be expected that a wetland will be producing target effluent values 2 or 3 years after completion of the planting, but this time is often waste and site specific.

Two periods exist when wetland planting is most successful: fall and spring. In the fall, tubers or clumps of aquatic emergent vegetation can be planted. Fall planting allows the plants to acclimate to the new soil substrate slowly, as wastewater is introduced at shallow depths. The hydroperiod (i.e. water depth and duration of flooding) of the wetland should stay below the tops of any newly planted emergent vegetation clumps or tubers. The other planting period is spring when seeds, sprigs, tubers, and/or clumps can be introduced. Water level control is much more critical to spring planting of sprigs, seeds, and tubers.

The most successful planting method for emergent vegetation, in either fall or spring, is by placing clumps of 4 to 10 plants into the wetland on 0.6 to 1.0 m staggered centers. These clumps include the native soils, along with multiple tubers, which ensures the highest success rate of wetland planting. This type of planting is limited to smaller systems and in areas where plant material is available for harvest. Backhoes and dump trucks can be used to extract and harvest the plants from acceptable and approved harvesting areas. Stems can be cut off in late summer and fall plantings to facilitate transporting and planting of the clumps. The cost of planting clumps is dependent on the distance to the source of material. It is possible to have a fully functioning wetland in 1 to 2 years after planting with emergent clumps.

Other planting techniques include the use of purchased tuber stock and seeds from commercial sources. Tuber stock is typically planted in a similar fashion to transplanting seedlings and, depending on the size of the stock, can be planted in spring or fall. For small tuber stock, spring planting is best. The use of seed is the most risky way to vegetate wetlands. Seed treatment (acid, base, oxidizing agents), seed placement (hand casting, hydroseeding), and water level manipulation are all critical factors in the success of seed germination. Planting with seed is less expensive than planting clumps or tubers, but the success and rate of vegetation development is much less.

When planting seeds, sprigs, or tubers, it is necessary to bring to water levels up slowly with the plant growth, starting at an initial depth of 20 to 30 mm, and slowly increasing depth to 200 to 300 mm as the plants grow. Slow water depth increases also ensure that wetland vegetation does not float before the roots take hold into the soil. If the size of the wetland does not allow a 0 to 0.1 percent bottom slope, then grading the wetland bottom into small sections separated by shallow internal berms (200 to 300 mm in height) will be required. This particular requirement is not needed when planting techniques incorporate clumps of soil, roots, and stems.

Planting of the wetland should be done as soon as possible in the construction sequence of a wastewater treatment plant. Often during wetland start-up, the water quality is degraded due to algal growth, sediment resuspension, and wildlife activity in the more open shallow water units. Treatment wetland designers, owners, and regulators all need to take into account the method and time of year of planting, as these are determinants for the time needed for wetland vegetation to mature and hence the startup period needed between initial planting and the production of effluent meeting discharge permit requirements.

Regional sources are usually able to supply relatively small amounts of plant material, which in turn may influence initial planting densities. Planting tubers or clumps on 0.5 m centers, for example, requires approximately 40,000 plants per hectare. Planting on 1 m centers require 10,000 plants per hectare. If given a planting constraint (cost or availability), it is better to place more plants in the last half of the cell, than in the first half. It is important to ensure success of planting of the last effluent half of the wetland. Planting 5 to 10 m wide vegetated strips across the wetland width and perpendicular to the flow will minimize short-circuiting, and allow for a future source of plant material for later planting.

The emergent plants of choice for wetland treatment purpose are *Scirpus species* (bulrushes) and *Typha species* (cattails). Of these two, *Scirpus spp.* appears to have higher treatment potential based upon surface area. Hardstem bulrush, for example, grows at higher stem densities, which affords much greater specific surface area in the water column than does cattails. This specific surface area is a critical growth location for attached microflora and microorganisms. Cattails are generally larger in diameter than the bulrush, and have a much larger stem to tuber transition in the water column. Further, hardstem bulrush does not contribute as much detrital material during the dormant period as cattails, thereby reducing potential BOD leaching back into the water column. *Scirpus spp.* wetlands have about one-third the background BOD as *Typha spp.* wetlands.

The seasonal change in plant community coverage in a FWS constructed wetland is shown for the City of Arcata's Enhancement Marshes in Table 6-1. The loss of open water to duckweed and sago pondweed coverage from spring to fall is evident from the data given in Table 6-1.

Wetland plant growth and survival is also dependent on environmental factors other than hydroperiod. Two of these include soil texture and soil chemistry. Many wetland plants grow rapidly in soils of sandy to loamy texture. Soils with excessive rock or clay material may retard plant growth and actually result in mortality. Excessively acidic or basic conditions may limit the availability of nutrients required for plant growth. In some cases, soil concentrations of macro or micronutrients may not be available in the native soil for initial plant growth, and organic fertilizers may have to be used.

TABLE 6-1
Percent of dominant plant species areal coverage of the Enhancement Wetlands of the Arcata Marsh and Wildlife Sanctuary.

	Enhancement marsh units, date					
	Allen Marsh		Gearheart Marsh		Hauser Marsh	
Type of Cover	April 1986	Sept. 1987	April 1985	Sept. 1985	April 1986	Sept. 1987
Open water	70.0	36.2	83.8	5.0	32.5	23.0
Common cattail		6.3	5.5	6.0	10.5	4.3
Marsh pennywort	5.6		10.0	11.8	27.0	
Sago pondweed		NV[b]		77.2	NV[b]	NV[b]
Alkali bulrush		11.9				0.8
Lesser duckweed		40.0[a]			30.0[a]	69.6
Hardstem bulrush						2.3
Common spikerush				0.7		
Upland grass spp.	30.0					

[a] Duckweed coverage was too low because the wind had pushed it into windrows.
[b] NV = not visible because of duckweed coverage

As discussed earlier, there are several water quality reasons for balancing the amount of open water area (submergent and floating), and the amount of vegetated water area (emergent). Dissolved oxygen levels are maintained at higher levels in open water areas, which supports aquatic organisms such as aquatic insect larva, amphibians, and fish.

Impermeable Barrier and Liner Materials

A major concern with FWS constructed wetlands is the potential loss of water from infiltration. While there are some wetland applications where infiltration is desirable, the majority requires some type of barrier to prevent exchanges with groundwater. Under ideal conditions, the wetland site will consist of natural soils with low permeability that restricts infiltration. However, many wetlands have been constructed or proposed on sites where soils have high permeability. In cases where waste and site specific conditions warrant, some type of liner or barrier can be required to restrict infiltration. Some general guidelines and specifications for minimizing infiltration and berm storage losses are as follows.

Existing natural site soils with permeability less then approximately 10^{-6} cm/s are generally adequate as an infiltration barrier. For site soils with higher permeability, some type of liner material is required. Some examples of wetland liner materials include bentonite soil layers, chemical treatment of existing soils, asphalt, and synthetic membrane liners. In some instances, existing in-situ soils can be compacted to acceptable permeability. Whatever liner material is chosen, an important consideration is to provide adequate soil cover and depth that protects the liner from incidental damage and root penetration from the wetland vegetation. Another consideration should be given to burrowing mammals such as muskrats, nutria's rats, etc., which can do substantial damage by chewing and consuming liner material.

Operation and Maintenance

The operation and maintenance of FWS constructed wetlands is much less demanding than for mechanical wastewater treatment technologies such as the activated sludge or trickling filter processes. Routine operation and maintenance requirements for wetland systems are similar to those for oxidation pond systems, and include hydraulic and water depth control, inlet/outlet structure cleaning, grass mowing of berms, inspections of berm integrity, wetland vegetation management, vector control, and accumulated solids/peat management if required.

Operation and maintenance considerations for FWS constructed wetlands are as important as design issues in meeting regulatory requirements pertaining to effluent water quality. The treatment effectiveness of most of the existing FWS constructed wetlands can vary considerably depending on water depth, weir overflow rate, plant density/plant location, and wildlife activity. Following are some of the more important operation and maintenance considerations for FWS constructed wetlands.

Management of FWS Constructed Wetlands

Limited attention has been paid to the overall operation and maintenance strategies of a FWS constructed wetland to meet water quality objectives. To be accepted readily by regulatory agencies and owners, more effort should be directed to developing holistic and sound management plans that cover a wide range of issues associated with FWS constructed wetlands (Hammer 1992). Many management issues pertaining to FWS constructed wetlands are not mutually exclusive. Typically, one management decision or action influences other management goals.

Listed below are considerations that need to be addressed when developing a FWS constructed wetlands management plan:

- regulatory requirements
- hydroperiod and hydraulic retention time—water depth and flowrate
- hydraulic control—weir overflow rate/Inlet-outlet distribution
- vegetation control (planting, harvesting, and monitoring)
- proximity of airports
- wildlife management
- vector control (mosquitoes)
- structural integrity of berms
- nuisance conditions (odors)
- inlet/outlet structures
- public access
- environmental education

A set of operation and maintenance procedures needs to be developed for each of the goals of the management plan developed above. This management manual should be organized in a manner to assist the operator and owner in effectively operating the wetland system under a wide range of environmental conditions. At a minimum, the following categories should be included for each goal of the management plan.

1. Objective and goal for the component
2. Startup condition/monitoring
3. Normal operating condition/monitoring/lead time
4. Abnormal operating condition/monitoring/lead time
    - Problems
    - Indicator
    - Cause of abnormal condition
    - Course of action to solve problem
5. Maintenance requirements
6. Sampling/monitoring program

Potential Nuisance Conditions

Constructed and natural FWS wetlands are typically enriched semi-natural wetland ecosystems. Because of their very nature, they have the potential to create conditions that may be a nuisance to human neighbors or to the wildlife species they harbor. Nuisances that could conceivably occur include mosquito breeding, creation of odors, attraction of dangerous reptiles (snakes and alligators), potential for accidental drowning, and the potential for bioaccumulation or biomagnification of pollutants in wildlife (Hammer 1992; Wass 1997). There is limited quantitative FWS treatment wetland data available for these potential nuisances; however, some information is available on mosquito and odor control. Unfortunately, there is inadequate data to date on any of these issues to help assess all possible effects when implementing a FWS treatment wetlands.

Wetlands and other stagnant water bodies can provide breeding habitat for mosquitoes. Some of these mosquito species can transmit diseases to humans or to valuable livestock. In addition, mosquitoes may be a nuisance because of their large numbers and painful bites. Few quantitative data have been published on mosquito population densities in treatment wetlands although a large number of treatment wetland systems are periodically monitored for mosquito larvae and pupae populations. General conclusions are that the numbers of breeding mosquitoes in treatment wetlands are not higher than in adjacent natural wetlands (Crites et al. 1995). When mosquito populations are present, their numbers appear to be directly related to organic loadings (Martin and Eldridge 1989, Stowell et al. 1985, Wieder et al. 1989, Wile et al. 1985, Wilson et al. 1987).

Generally, odors in FWS treatment wetlands are associated with high organic loadings, especially in the inlet region of the wetland. It has been observed that most treatment wetlands have odors similar to the normal range of odors observed in natural wetlands. No published qualitative information has been found during preparation of this assessment on odors associated with treatment wetlands.

Dangerous reptiles, including poisonous snakes and alligators, are attracted to FWS treatment wetlands in some regions of the U.S. These same species are generally a natural component of natural wetlands in those same areas, and most citizens are aware of the need to avoid these animals when they are encountered. No published information has been found on population densities of these organisms in treatment wetlands or relating the occurrence of these species to wetland design. Further, no data has been found indicating that treatment wetlands are more or less likely to create risks to wildlife species than adjacent natural wetland ecosystems. This issue is being examined further through another EPA-funded project in progress.

Vegetation Management Implications

Routine harvesting of vegetation is not usually necessary for FWS constructed wetlands (Reed et al., 1995). In many cases, the only routine vegetation management consists of annual or biannual harvesting of emergent vegetation from designed open water areas, and inlet/outlet structures. Over some period, whose exact length is unknown, some removal of accumulated plant material and detritus may be required in FWS constructed wetlands. Studies at Arcata have indicated that detrital/litter has reduced the wetland volume by about 50 percent in 12 years with no apparent change in performance. However, harvesting may be required if vegetation growth cycles significantly reduces pollutant removal efficiencies, restricts water flow, affects habitat goals of the project, and or inhibits wetland operation and maintenance activities.

If routine harvesting is required, it is recommended that the vegetation be removed in 5 to 10 m strips perpendicular to the direction of flow. This strip should be replanted and allowed to grow, before the next adjacent strip of vegetation is harvested. This process can be repeated over a number of years. The primary goal of this type of vegetation harvesting is that the wetland is never completely devoid of vegetation at any one time. The harvested vegetation can be transplanted, composted or burned; harvested wetland vegetation has also been used for the production of methanol. It is also important to consider potential effluent water quality impacts during vegetation harvesting. Typically, the wetland cell being harvested is taken off line during this time.

One problem that can be very difficult to manage for is the potential for animals, in particular nutria and muskrat, to use the emergent wetland vegetation as a food source. Some FWS wetland systems have had these creatures consume all the emergent vegetation. If this occurs, the only action possible is to trap and relocate the animals, and re-vegetate the damaged wetland cells.

Mosquito Control

Mosquitoes are common in any wetland or open water environment. However, in some cases, especially urban environments, a FWS constructed wetland can produce mosquito populations that are viewed as a nuisance by the public or pose a true public health threat. Mosquito populations appear to be controlled effectively in FWS treatment wetlands by small fish, such as the mosquito fish (*Gambusia affinis*) (Dill 1989, Steiner and Freeman 1989). However, fish may not be able to control mosquito populations in portions of FWS treatment wetlands that are colonized by dense populations of floating vegetation mats (Walton et al. 1990). This condition can be avoided by designing the FWS constructed wetland with open water areas and by paying attention to vegetation densities in emergent areas. Other animals, such as frogs, birds, and bats, may also contribute to controlling mosquito populations.

Although biological methods have, and continue to, show promise for controlling mosquitoes in treatment wetlands, mechanical means are also available which may complement these efforts. Sprinklers have been successfully utilized to control adult mosquito populations in constructed wetlands (Epibare et al. 1993). The spray from overhead sprinklers disrupts the water surface and affects the ovipositioning. This technique was very effective in reducing mosquito larva production in a FWS wetland; however, additional capital investment in the spray equipment and operation and maintenance of the pump and sprinkler system is required.

Bacterially derived larvicides are another available mosquito control option. As with any agent, whether it is a fish, another invertebrate, or a larvicide, the effectiveness depends upon getting that agent in proximity to the target organism, in this case mosquito larvae. This may require combining vegetation removal with an efficient means of broadcasting, or otherwise delivering the larvicide such that adequate basin coverage is achieved. The two most common mosquito larvicides available are derived from *Bacillus thuringiensis* (Bti) and *B. sphaericus* strains. When adequate basin coverage is achieved, both agents have been used effectively to control mosquito populations. Bti was applied to the Sacramento Regional demonstration wetland cells when mosquito larva reached 0.1 larva/dip. Bti was applied to an entire half-cell at a rate of approximately 2 kg (liquid) per hectare. Repeated Bti application and vegetation harvesting around the edge appeared to be effective in avoiding high larval densities. Vegetation harvesting and application of 11.2 kg/ha (10 lbs./acre) of granular *B. sphaericus* on roughly 3-week intervals during the summer of 1999 at the Tres Rios demonstration wetlands also resulted in low mosquito larval counts and appeared to be effective at lowering adult populations.

Process Control

FWS constructed wetlands have minimal need for active process control. The only two operational controls for FWS wetlands are hydraulic loading and outlet weir level control (if designed to allow varying hydro periods). Further, hydraulic loadings can only be varied if alternative hydraulic pathways exist.

Under certain conditions, increasing the outlet weir level for a given period of time will result in no discharge. This would allow for short-term periods of no discharge to a receiving system. This increase in water level will increase the HRT while maintaining the areal loading at a constant value. The maximum depth that can be tolerated by emergent plants in the FWS wetland limits the degree of water level increase. Generally this maximum depth is 1.0 to 1.5 meters while a more normal range for emergent plants is 0.4 to 0.75 meters.

Monitoring Requirements

The most critical monitoring issue during the wetland startup period is vegetation growth and coverage. A wetland that does not develop sufficient emergent/submergent vegetation becomes a shallow oxidation pond, producing algae, BOD, and solids. The planting strategy, combined with hydroperiod control as the plants grow, determines the effectiveness of vegetation growth during the startup period. Other monitoring factors include control of aquatic birds, mammals, and invasive vegetation during the startup period.

Once the wetland vegetation has been established, the system can be brought on line and wastewater introduced. After the startup period is over, routine-monitoring requirements will often be necessary. Most important in the operation of a FWS constructed wetland is monitoring hydraulic and organic loadings to, and discharge(s) from, the wetland system (including the monitoring of individual wetland cells). Such tasks require measuring influent and effluent flowrates, and recording of water depths in each wetland cell. This information has not been collected routinely from many existing FWS constructed wetland systems and that has slowed the broad-based acceptance of this technology because data such as these can be used to assess inlet/outlet distribution and performance. It is surprising that many wetland treatment systems were not designed to gather this type of data even though this information can be used to develop seasonal strategies, based upon hydraulic and organic loadings, hydraulic detention times, and areal loadings.

Influent and effluent water quality constituents should also be measured on a weekly or, at minimum, on a monthly basis. Parameters such as BOD, TSS, pH, nutrients, temperature, specific conductance, and dissolved oxygen should be monitored as these parameters can be used to assess wetland performance, and determine constituent loadings. Table 6-2 lists suggested monitoring tasks for a FWS constructed wetland; these data are important for understanding the system performance and would improve the state-of-the-art for future design efforts.

TABLE 6-2
Suggested monitoring requirements for a FWS constructed wetland.

Monitoring requirement	Location of monitoring	Frequency of monitoring	
		Large system	Small system
Hydraulic monitoring			
Water depth	Each cell	Weekly	Weekly
Inlet flowrate	Inlet of each cell	Daily	Weekly
Outlet flowrate	Outlet of last cell	Daily	Weekly
Water Quality monitoring			
Dissolved oxygen	Inlet each cell, outlet last cell	Weekly	Monthly
Temperature	Inlet each cell, outlet last cell	Weekly	Monthly
Conductivity	Inlet each cell, outlet last cell	Weekly	Monthly
pH	Inlet each cell, outlet last cell	Weekly	Monthly
BOD	Inlet each cell, outlet last cell	Weekly	Monthly
TSS	Inlet each cell, outlet last cell	Weekly	Monthly
Nutrients (e.g. TN, NH_4, NO_3, TP)	Inlet each cell, outlet last cell	Weekly	Monthly
Wetland biota monitoring			
Vegetation coverage/distribution	Each cell	Bi-annually	Annually
Wildlife (nuisance animals)	Each cell	Bi-annually	Annually
Vectors (mosquitoes, etc)	Each cell	Weekly during season	Weekly during season
Fish	Each cell	Monthly	Monthly
Birds[1]	Each cell	Monthly	Monthly
Aquatic insect larva[1]	Each cell	Monthly	Monthly
Civil Issues			
Berm and liner (if used) condition	All berms	Monthly	Monthly
Inlet/outlet condition	All inlet/outlet structures	Monthly	Monthly
Access road condition	All roads	Monthly	Monthly
Solids/peat buildup	Each cell	Annually	Annually
Public Use[1]			
Trail/sign conditions	All trails	Annually	Annually
Number of people	Access points	Annually	Annually

1. If required as part of management plan

Considerations for Minimizing Variability in Effluent Quality

Many items, most of which have been discussed throughout this Technology Assessment, combine to influence the variability of effluent quality from FWS constructed wetlands. The following are important design and operational considerations, which can influence and help control variability of effluent water quality, that need to be considered throughout the FWS constructed wetland planning and design process.

1. Ability to buffer weekly fluctuations in effluent flow by use of multiple cells.
2. Ability to store water individually in each wetland cell to allow for longer hydraulic detention times for BOD and TN removal, and quiescent conditions for settling processes.
3. Minimize the amount of emergent vegetation necessary to reach treatment goals. The aquatic vegetation contributes to background BOD, ammonia, and dissolved phosphorus levels in the wetland. The lower the influent BOD, TSS, and TN, the greater potential contribution this background source has to the variation in the effluent value.
4. If wildlife habitat is one of the goals of the project, it is important to have 3 to 7 days detention time of emergent vegetation at the final wetland outlet. This emergent vegetation zone of the wetland has minimal habitat value for migratory and resident birds (source control), and provides a final clarification/vegetative filter zone.
5. Design outlet collection zones, and inlet/outlet structures to minimize open water areas, which can attract wildlife and promote phytoplankton and periphyton production.
6. Minimize the velocity fields at the inlet and outlet zones of the wetland.
7. Design for solids removal at the inlet region of the FWS constructed wetland.

Research Studies

Treatment wetland research studies should be designed to answer specific, design-related questions. The size of wetland research cells, their source of feed water, water depth controls, and sampling can all affect the ability to scale up the conclusions to a full-size treatment wetland. Extremely small FWS wetlands may have edge effects that result in behavior that is unrealistic compared to full-scale wetlands. A pilot system may receive water in batch loads or in a nearly continuous mode, neither of which is typical of most full-scale treatment wetlands. Inlet constituent concentrations may be more constant in some pilot studies than can be expected with a full-scale system. In small pilot wetland cells (mesocosm-scale), sampling including plant harvesting can alter performance significantly.

Additional, long-term, well-funded research studies would be very valuable for advancing the FWS wetland technology. The Listowel, Ontario, database represents a major contribution for the development of design criteria and operational performance estimates for a cold-climate, cattail-dominated, constructed treatment wetland. Research studies have been performed on the City of Arcata, California, research wetland cells since 1980, with two major data reports 1983 and 1986, and several papers summarizing research activities and findings. The 4-year time frame of this research project and the excellent monitoring and data reports were essential for maximizing the research benefits of this project. It is recommended that regional sponsors be solicited to contribute additional data, following the example established by the Arcata and Listowel projects. Several facilities are currently available to provide a cost-effective basis for additional pilot research. These facilities include the Everglades Nutrient Removal test cells, the Champion pilot wetlands in Pensacola, Florida, the Tres Rios demonstration wetlands in Phoenix, Arizona, the Arcata, California, pilot wetland cells, the Orange County Water District, California, demonstration wetland, and the Eastern Municipal Water District pilot wetland cells in Hemet, California.

A common goal of all new and continuing wetland research studies should be the achievement of a high level of quality assurance for all data collected. Water flow and field parameters should be measured using calibrated instruments, and analytical tests should follow accepted testing methods with adequate quality control. All data should be validated prior to analysis and publication. Following good scientific research practices will help to improve our understanding of the transformation processes and will reduce the level of uncertainty in predicting treatment performance of free water surface wetlands in the future.

As part of an ongoing research effort, an interactive communication link for operators, owners, designees, and regulators of FWS constructed wetland should be established. The success of FWS constructed wetlands are not only dependent on good designs but dependent on good operators and management. A forum for discussion of operational issues with treatment wetlands will ensure the continued success of this important wastewater treatment technology.

Critical Research Issues

A number of critical research issues requiring additional information and data analysis have been identified in this technology assessment report. These issues deal with the relationships between design variables and system performance. Some of the more pressing issues requiring resolution include the following:

- Specific studies on effluent water quality as a function of characteristics of the inlet and outlet weirs, including number, location, type, spacing, overflow rate, and outlet capacity

- The effect of aspect ratio on internal flow patterns and wetland treatment performance
- Determination of volatile solids destruction and subsequent reuse of soluble biodegradable by-products in the water column
- In depth monitoring of selected full-scale projects to acquire an understanding of systems receiving medium to high organic loads, and of systems in cold climates
- A qualitative description of the factors affecting the high variability of removal rate coefficients
- Detailed data sets for calibration of the sequential performance equations used to describe the nitrogen transformations in FWS treatment wetlands
- The relationship between the settled/decomposing solids and the removal of BOD and ammonia in treatment wetlands
- The spatial distribution of solids removal and nitrogen transformation processes to identify wetland configurations and conditions that optimize performance
- The appropriate use of volumetric removal rate coefficients for treatment wetland data analysis given their dependence on loading rates and water depth
- The importance of dissolved oxygen concentrations in control of wetland performance for BOD and TN removal
- The effect of open/deep water zones on internal flow patterns and treatment performance
- The importance of design criteria such as plant selection and open/deep water zones on wildlife populations in treatment wetlands
- The effects of different plant communities on treatment performance for all major constituents of concern
- The role of substrate surfaces in support of epiphytes and their role in conversion and transformation processes
- Additional information on factors affecting metal removal in treatment wetlands including mass balances over extended operational time periods
- The normal range of quantitative fates and effects of potentially toxic metals and organics in treatment wetland biota
- Normal populations of levels of mosquitoes in treatment wetlands and an understanding of the physical, chemical, and biological factors affecting these populations

- Studies directed at the use of Integrated Pest Management (IPM) for managing mosquito populations in FWS constructed wetlands

Other important issues are more difficult to study in a single research effort, but instead need the collective input from wetland designers and operators of full-scale treatment wetlands. These issues include:

- The optimum design and management of wetlands for multiple uses such as treatment, habitat, and recreation
- The role of dissolved organic carbon generated from the decomposition of detritus in treatment wetlands
- The effect of managing the hydroperiod over a weekly and monthly period on the performance of treatment wetlands
- The role of the full range aquatic microorganisms and aquatic insect larva as they interact with the particulate material (public health significant organisms, plant litter, TSS, etc.) in a FWS constructed wetland

Finally, much effort remains to be done with State and Federal agencies in terms of defining the role and functions of FWS constructed wetlands in the various wetland policies, and in the development of appropriate discharge standards.

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