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WETLAND HYDROLOGY AND HYDRAULICS: WATER QUALITY ASSESSMENT

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ABSTRACT: Wetlands function in the transitional environment between terrestrial and aquatic ecosystems and their impact on water quality enhancement (as well as biological diversity) is directly dependent on the hydrologic, or hydraulic factors controlling the residence, or detention time of the water within the wetland. Similarly, constructed wetland performance, in terms of wastewater treatment efficiency, also depends on the processes affecting water detention within the system. In many cases, however, both the hydrologic, or hydraulic factors and the actual degradation processes or mechanisms associated with particular constituents of concern are only poorly understood. Successful wetland restoration, or design of constructed wetlands for wastewater treatment requires a thorough understanding of the site hydrology, or flow hydraulics as well as an assessment of the specific constituent degradation mechanisms of concern. In this paper, some of the current literature associated with development of the concepts outlined above is reviewed. Examples of how these concepts may apply to a range of wetland types are considered; including a sub-alpine bog/fen, a tidal marsh, and constructed surface and subsurface flow wetlands.

INTRODUCTION

Wetland functions and value in the landscape are now well recognized. Although rapid and considerable progress has occurred towards gaining a greater understanding of wetland ecosystems and their effects on water quality, unfortunately, much of the work remains fragmented not unlike natural wetlands themselves.

Wetland ecosystems may be broadly classified as freshwater or coastal wetlands. Both share a common feature of being the transitional area between terrestrial and aquatic systems. The biological productivity of these transitional landscape systems typically far exceeds either the terrestrial or aquatic systems in terms of species diversity and general abundance. It is this diversity and productivity that characterize their "robustness" and enable wetland systems to dramatically affect water quality. The general productivity, however, is directly linked to the spatial and temporal distribution of rates of water movement through the wetland (i.e., its "hydrology") and subsequent water quality throughout the ecosystem. "Hydrology" is only one of the three indicators typically used for wetland delineation along with hydrophylic vegetation and hydric soils. Clearly, the hydric soils play a role in the wetland water quality as the wetlands may function as potential contaminant sinks from upland terrestrial sources, or as potential contaminant sources to adjacent aquatic systems depending on the integrity of the wetland ecosystem.

Given their central role in regulating water quality in the landscape, wetlands are now considered beneficial natural resources whose ability to assimilate nutrients and degrade some

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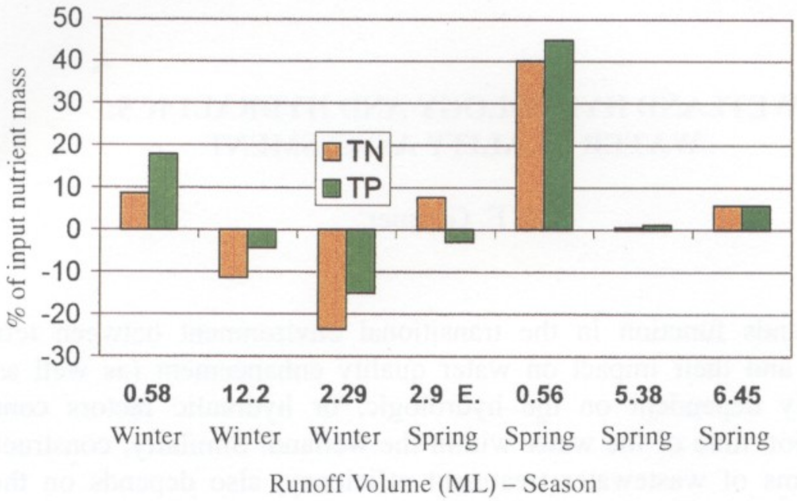

Fig. 1 Crooke's Wetland – Relative gain(+) and loss(-) of nutrients for individual storms

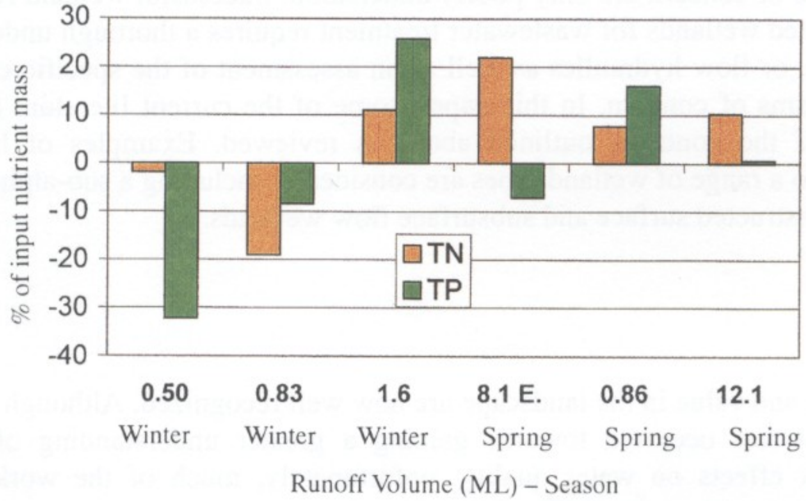

Fig. 2 Humphrey Wetland – Relative gain(+) and loss(-) of nutrients for individual storms

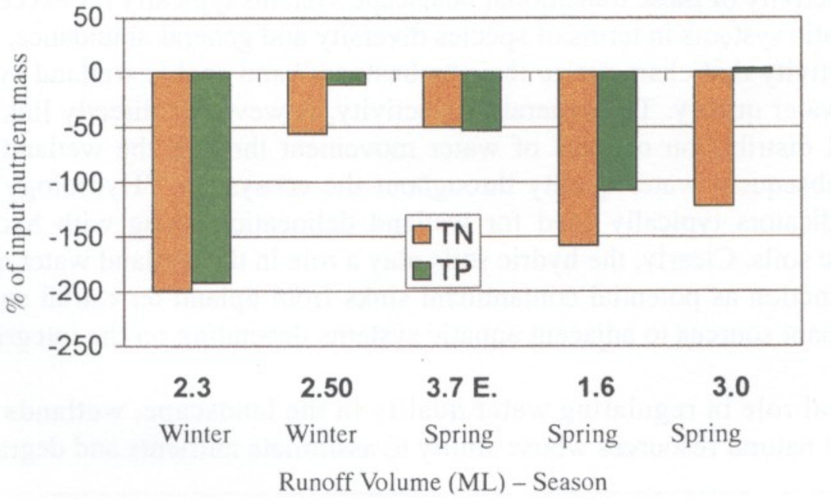

Fig. 3 Reid's Wetland – Relative gain(+) and loss(-) of nutrients for individual storms

opportunities remain for integrating treatment wetlands into the agricultural landscape so as to minimize adverse impacts of non-point pollution runoff on aquatic systems (Peterson 1998); similar in concept to the use of vegetative filter strips to minimize sediment, nutrient pesticide runoff into surface waters (Evans et al. 1997; Watanabe 1998).

Wetland functions are the physical, chemical and biological processes, or attributes that are vital to the integrity to the wetland/upland landscape interrelationships. Some wetland values or attributes, though not essential to the integrity of the landscape, are perceived as important to the public. However, some wetland ecotones act as simple transformers of nutrients or contaminants that are temporarily detained as they pass through the wetland and are later released into the water column as water levels fluctuate. For example, in a study of Australian wetland nutrient inputs to the environment, Crooke's and Humphrey wetlands in Figs. 1 and 2 took up much of the nutrient load while Reid's wetland in Fig. 3 tended to release nutrients due to preponderance of groundwater springs within the wetland. As yet, there is insufficient mechanistic understanding of particular wetland biogeochemical processes functioning (Reddy and Gale 1994) in the soil-plant continuum and the role of these processes in chemical retention and release. In addition, the hydrologic/hydraulic characteristics of the wetland often determine the detention times available for contaminant degradation/retention, as well as, redox potentials in the soil environment. Moreover, long-term chemical retention in wetlands is complicated by accretion of new sediments and settling of particulates (Kadlec 1992). Clearly, the rate of sediment/particulate accretion and potential removal in the wetland depends on the water flowpaths and velocities through the wetland.

Similar to integrated models developed for vadose zone processes and groundwater contamination (Grismer et al. 1997), an integrated biogeochemical-hydrologic model for particular chemicals of interest may be required to adequately manage water quality aspects of wetland systems. With sufficient information, such a model could be linked to the biological ecosystems supported by the wetland. That is, the mechanistic detail could be incorporated into the ecosystem modeling approach identified by Mitsch (1992). In the following section, a review of some of the current literature considering aspects of wetland modeling, hydrology, hydraulics, nutrient uptake and specific constituent impacts on water quality is undertaken with the goal of highlighting some of the linkages necessary for integrated modeling of wetland water quality.

WETLAND ECOSYSTEM MODELING

Modeling the complete wetland ecosystem depends on the type of wetland system under consideration (e.g., freshwater vs. tidal marsh) and the objective of the modeler (e.g., nutrient/contaminant fate, predictive, or design models vs. generalized system management). Preliminary modeling considerations of wetland ecosystems were focused on the basic hydrologic setting of the wetlands and their potential biological communities, or their successional stages, with the emphasis on restoration efforts. As the literature concerned with use of constructed wetlands for wastewater treatment developed and expanded (particularly in the engineering-related literature), more attention was directed at providing detailed descriptions of some of the particular wetland processes.

Presently, efforts to model wetland systems can be broadly classified as:
(1) GIS-based or landscape wetland management models;
(2) general hydrologic/hydraulic routing models;
(3) biological community diversity/succession models; and
(4) chemical/contaminant transport/fate models.

These modeling categories can also be considered in terms of their spatial or temporal resolution. For example, GIS-type models generally consider the wetlands as a whole, or as part of the watershed landscape (e.g., Vadas et al. 1995) and can be linked to water quality assessment (Still and Shih, 1991). Similarly, biological succession-type models may also consider the landscape spatial scale, but generally consider temporal scales on the order of years to centuries. Though wetland hydrologic conditions may change over decades, present modeling efforts generally consider the present time frame and seasonal, or event-driven hydrologic events and their effects on water movement through the wetland. Finally, chemical fate models may be site and media (i.e., soil, water, plant or air) specific within the wetland for processes that occur fairly rapidly within hours to days (e.g. carbon, nitrogen and phosphorous cycling, or contaminant adsorption/uptake). Modeling efforts that consider sediment accretion, or bog/fen formation processes and their effects on water quality within the wetland tend to cross both of the spatial and time scales considered above.

Successful wetland ecosystem modeling first considers the geohydrologic setting of the wetland (e.g. Figs. 1 - 3), then the biological community that the wetland supports and finally, the interrelationships between the two that determine the water quality characteristics of the wetland. Landscape conceptual models that characterize the hydrologic setting of freshwater (Bedford 1996), estuarine (Turner and Rao 1990) wetlands, as well as the geohydrologic setting of wetland systems are under development (e.g., Cole et al. 1977) and will be useful to set the stage for general ecosystem models. Existing hydraulic or hydrodynamic models exist that can be adapted for use in wetland systems including hydraulic routing models (Guardo and Tomasello 1995; Konyha et al. 1995), combined routing-agricultural drainage models (Giraud et al. 1997), and tidal system models (Suhayda, 1997; Hsu et al. 1998). Hydraulic-routing type models (e.g. HEC-RAS by the US Army Corps) are important towards evaluation of water velocities, or detention times within the wetland soil and water column. Identification of the spatial and temporal (seasonal) distribution of such times is essential towards determination of the rates of sediment deposition and chemical fate and transport, that is, the water quality within the wetland. Existing vadose zone models of water flow, chemical transport/detention, and plant root water and nutrient update (Simunek et al. 1994) can be linked to the hydraulic surface water models. In the model linkage, the many possible interrelationships between different phases of the wetland system must be considered such as that between the vegetation flow velocities and sediment suspension/deposition (Brueske and Garret 1994). Detailed biogeochemical cycling/uptake considerations such as described by Rolston and Jayaweera (1992) for nitrogen and effects of sediment re-suspension on phosphorous and nitrogen uptake described by Mitsch and Reeder (1991) and McIntyre and Riha (1991) are also important to incorporate into model development. Detailed trace element transformation/detention models linked to wetland soil redox conditions can be included as they are developed from constructed wetland applications for treatment of particular wastewater streams. For example, data is available describing ammonia and metals removal from municipal effluent treatment wetlands (Crites et al. 1997), retention of Al and Fe from acid-mine drainage (Mitsch and Wise 1998), and degradation of hydrocarbons (Machate et al. 1997; Salmon et al.1998) and some herbicides such as atrazine and its derivatives (Mersie and Seybold 1996). Finally, as noted initially, the geochemical setting of the wetland ecosystem cannot be overlooked in terms of its impact on wetland water quality and related vegetation.

WETLAND ECOSYSTEM HYDROLOGY

The wetland hydroperiod, whether naturally occurring, or managed as in constructed wetlands, and its source of water (wastewater, sea water, surface runoff, or groundwater) are often the defining characteristics of the wetland ecosystem and its water quality. This is particularly evident in tidal marsh systems in Fig. 4. Several studies have been conducted that consider the basic hydrologic processes for different wetland systems (e.g., Gehrel and Mulamoottil 1990) including arctic systems (Woo and Dicenzo 1989; Woo and Waddington 1990; Rovansek et al. 1996; Buttle and Fraser 1992). However, most recent studies have focused on the role of groundwater in wetland systems. Quantifying the rates of water transfer between the surface and subsurface are possible using standard hydrologic techniques and may be critical to evaluation of wetland health and function. For example, Hayashi et al. (1998) determined rates of water and solute transfer between a prairie wetland and adjacent uplands in Saskatchewan, Canada through tracking chloride in the system. They found that snowmelt runoff was the dominant water input to the wetland and it transported 4-5 kg/yr of chloride from the upland to the wetland. After infiltrating under the wetland, the chloride moved laterally with shallow groundwater to the upland areas. Once below the upland, chloride moved upward with soil water that is satisfying evapotranspiration demand and accumulates near the surface. Part of this chloride mixes with snowmelt runoff and returns back to the wetland such that the chloride is cycled between the wetland and the upland. While this chloride cycle occurs within 5-6 m of the ground surface, a small amount of chloride escapes downward into a deeper aquifer. The estimated flux of chloride leaving the cycle is 0.1-0.6 kg/yr, or about the same order of magnitude as the rate of atmospheric deposition of chloride. Thus, knowing the rate of atmospheric input of chloride enables use of the concentration of chloride in groundwater under recharge wetlands to estimate the recharge rate of deep aquifers. In a similar system, Arndt and Richardson (1989) found that the spatial development of brine chemistry in a prairie-pothole wetland system in North Dakota results primarily from the successive mineral precipitation in groundwater being evapo-concentrated while flowing in the recharge-throughflow-discharge marsh/meadow complex following spring snowmelt. Roulet (1990) found that groundwater was the primary source of water maintaining a small headwater drainage basin swamp in southern Ontario. Groundwater flow from a regional aquifer as well as diffuse spring flows into the forested swamp exceeded rainfall by an order of magnitude and were important to the biogeochemical transformation of the emerging groundwater leaving the basin. Similarly, Gilvear et al. (1993) measured a large,

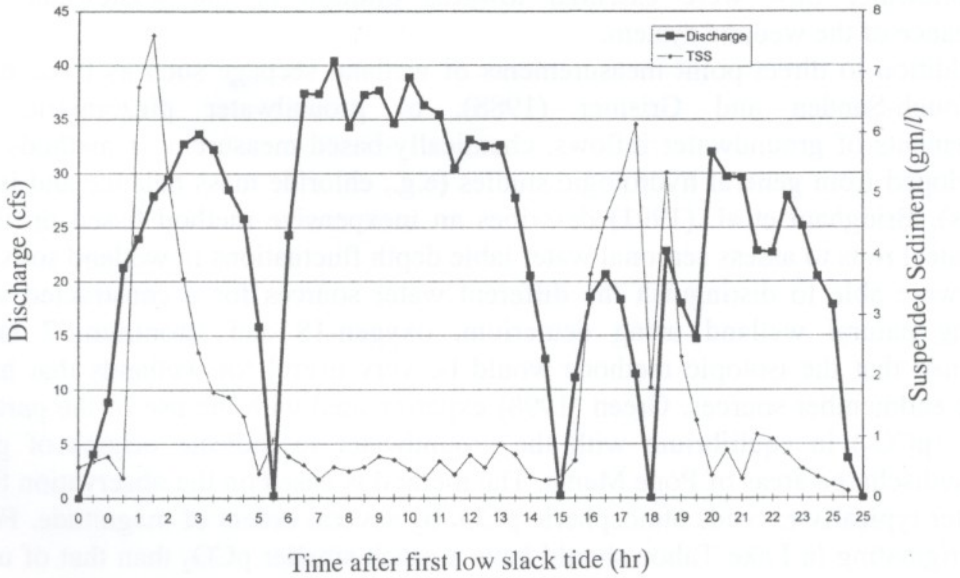

Fig. 4 Tidal marsh – western drainage system flood-ebb cycle (8/7-8/98)

vertically-upward oriented groundwater piezometric head-gradient arising from a unique local geologic formation that resulted in the groundwater contribution to the fen/wetland being the dominant input. The calcium bicarbonate rich groundwater precipitates tufa as if surfaces and leaves the calcareous fen via surface drainage channels. Even in a tidal marsh system, Nuttle and Harvey (1995) found that groundwater inputs comprised over 62% of the marsh inflows and directly impacted marsh water quality characteristics.

The importance of quantifying rates of groundwater movement cannot be overemphasized. De Mars and Garritsen (1997) used detailed groundwater modeling combined with field measurements to determine the groundwater flow patterns in a small wetland reserve of the Netherlands and found that groundwater flows were primarily lateral below the wetland rather than vertical as had been assumed. This determination was critical towards alleviating concerns about the limited conservation and restoration potential of the farm meadows. Seasonal aspects are also important. Glenn and Woo (1997) noted the temporal variation in groundwater input to a valley-bottom wetland on Ellesmere Island from spring to summer. Similarly, Green (1998) using direct measurements and mass balance methods found that snowmelt and surface runoff dominated the water balance and water quality of the sub-alpine Pope Marsh adjacent to Lake Tahoe during the winter/spring seasons. During the summer months, however, lateral groundwater inflows from the Lake combined with evapotranspiration were the primary water balance terms.

In constructed wetland systems, groundwater flows are also critical, but they are often determined as the closure term from mass-balance considerations. For example, Nolte and Associates (1997) describe a tertiary-treatment free-water surface wetland designed to remove heavy metals, trace organics and other constituents from municipal effluent in which the seepage loss to groundwater averaged 39% of the 1.23 mgd inflow. No direct measurements of seepage losses, or their impacts on groundwater were made though no adverse impacts were anticipated. Similarly, Mitsch and Wise (1998) were unable to account for 13% of the inflow into a wetland designed for metal removal from an acid-mine drainage stream and attributed this loss to seepage on measurement errors. In contrast, Brown and Stark (1989) found that for a treed fen combined with a cattail marsh designed for tertiary treatment of municipal wastewater built on a glacial outwash resulted in 44% and 21% groundwater inflows into the fan and marsh, respectively, with only surface water outflow. These fractions of groundwater flow were essential towards dilution of the wastewater stream and maintenance of the wetland system.

In addition to direct point measurements of wetland seepage such as those described by McCullough-Sanden and Grismer (1988), or groundwater piezometric head-based measurements of groundwater inflows, chemically-based measurement methods continue to be developed from general hydrologic studies (e.g., chloride mass balance and isotopic ratio methods). Bridgham et al. (1991) describes an inexpensive method based on the depth of rusting steel rods to assess seasonal water table depth fluctuations in wetland soils. Hunt et al. (1998) were able to distinguish the different water sources for a constructed wetland and adjoining natural wetland using deuterium, oxygen-18 and strontium-87 isotopes and determined that the isotopic methods would be very useful for wetlands that have distinct isotopic endmember sources. Green (1998) experimented with the use of the partial pressure of CO_2 (pCO_2) in equilibrium with the groundwater to indicate origins of groundwater recharge/discharge areas of Pope Marsh. The method is based on the observation that pCO_2 of soil-water typically exceeds atmospheric pCO_2 by several orders of magnitude. For example, water originating in Lake Tahoe should have a much smaller pCO_2 than that of upland soils, especially marsh soils in which active microbial respiration processes are underway. He found that the spatial distribution of pCO_2 was consistent with an extensive hydrogeologic investigation of groundwater recharge/discharge areas within the marsh but that interpretation of the pCO_2 was often confounded and sometimes inconclusive. Perhaps as important as identifying and quantifying the water inflows/outflows from wetland systems, is some characterization of the flowpaths through the system - such characterization is critical towards assessment of changing water quality conditions in the wetland system.

WETLAND FLOWPATHS

Characterization of water flowpaths through wetland systems is usually based on tracer studies. The tracer studies are then evaluated in terms of chemical reactor analogs as either simple plug-flow, completely mixed, or combination systems (Kadlec and Knight, 1996). For example, Nolte and Associates (1997) used a LiCl tracer and found that their free surface constructed wetlands operated primarily as plug-flow reactors with some mixing despite significant tracer losses in two of the experiments. In contrast, King et al. (1997) and Grismer et al. (1998) found that subsurface flow constructed wetlands typically function as mixed flow systems, or systems with substantial dispersion. Perhaps more significant, is that both studies indicated considerable non-uniform flowrates through different parts of the wetlands. For example, consider the detention time (DT) variation in Fig. 5 within the constructed wetland system of Grismer et al. (1998). Similarly, Stairs (1993) measured the hydraulic efficiency of free-water surface constructed wetlands for treatment of paper pulp mill effluent and found significant "short-circuiting" and stagnant zones within identical vegetated, non-vegetated and rock-filter cells. For the planted cells, the effective treatment volume was only 15-25% of the full volume due to stagnant zones and that early arrival of tracer distribution peaks by 30-80% of theoretical detention times indicated flow "short-circuiting". Grismer et al. (1998) also measured reduction of actual detention times of 15% and 30% in non-vegetated and vegetated subsurface flow systems, respectively, and that flow occurred primarily through the rootzone of the vegetated system and along one side of the non-vegetated system. Knowledge of the changing flow path conditions within the wetland system combined with the changing chemistry can be used to assess impacts on the wetland supported biological communities.

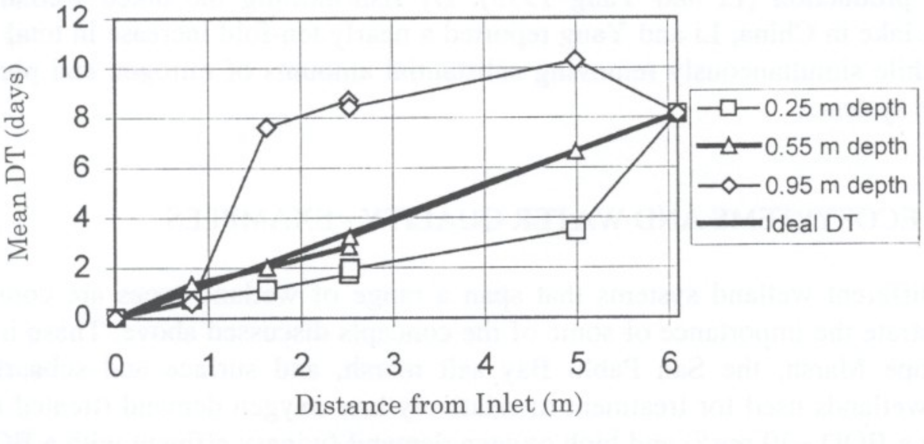

Fig. 5 Variation of hydraulic detention time within subsurface flow constructed wetland

WETLAND HYDROLOGY AND BIOLOGICAL COMMUNITIES

The interrelationships between the vegetative/biological communities associated with wetland ecosystems and the wetland water quality are very difficult to assess and often seemingly conflicting. Harbor (1994) developed a simple practical means of assessing impacts of changes in land use on basic wetland hydrology that may be useful towards evaluation of impacts on biological communities. Ehrenfeld and Schneider (1993) studied mature white-cedar swamps in disturbed and undisturbed watersheds of New Jersey to determine the relationships between changes in water quality and hydrology and species composition and community structure. As compared to the undisturbed watershed swamps, developed watershed swamps supported a larger fraction of facultative upland species and a larger proportion of non-hydrophytic vegetation. Changes in species composition, evaluated in terms of increasing fractions of invasive species, could be correlated with the number of perturbations in wetland hydrology and water quality indicators, but not their absolute magnitudes. Childers and Gosselink (1990) evaluated disturbance of forested wetlands in Louisiana in terms of cumulative impacts on the sediment loading, and nitrogen and phosphorous composition of the water. As expected, clearing of the forests resulted in increased sediment loading, turbidity, total nitrogen and phosphorous and temporal trends in nutrient composition of the sediments suggested that water quality had been declining for at least four decades. Shepard (1994) found similar affects of forest management activities on forested wetland water quality in terms of diminished water quality due to forest clearing. These adverse water quality impacts can be partially reversed and assessed in restored riparian wetlands (Vellidis et al. 1993). Greiner and Hershner (1998) examined the effects of landscape position and land-use on phosphorous retention in sediments of small coastal watershed wetlands of Virginia and were unable to find a relationship between them. In contrast, several researchers appear to have been able to relate vegetation or tree growth to changes in water quality and hydrology. For example, Niswander and Mitsch, (1995) modeled a created riparian wetland in Ohio, while Wallace et al. (1996) and Keeland et al. (1997) considered general tree growth response to changing hydrologic regimes. Similar concepts apply to coastal wetlands (Coats et al. 1989). Grismer et al. (1998) found that restoration of tidal flushing in a salt marsh of the San Francisco Bay region resulted in natural reestablishment of the pickleweed vegetation within the first two years of improved drainage and tidal circulation (see Figs. 6 and 7). The wetland vegetation hydrologic regime and water quality also directly affects the benthic microbial community (Boon et al. 1996) and aquacultural production (Li and Yang 1995). By fish-farming the diked wetland created adjacent to a lake in China, Li and Yang reported a nearly ten-fold increase in total fish yield and profit while simultaneously removing substantial amounts of nitrogen and phosphorous from the lake system.

WETLAND ECOSYSTEMS AND WATER QUALITY - EXAMPLES

Here, four different wetland systems that span a range of wetland types are considered in order to illustrate the importance of some of the concepts discussed above. These include the subalpine Pope Marsh, the San Pablo Bay salt marsh, and surface and subsurface flow constructed wetlands used for treatment of relatively low oxygen demand (treated municipal effluent with a BOD ~30 mg/*l*) and high oxygen demand (winery effluent with a BOD ~5000 mg/*l*) wastewaters, respectively. In these examples, the importance of evaluating the wetland ecosystem setting, adequate characterization of the water source and its dominant flow path, the interrelationships between wetland vegetation and water quality, and finally, a very brief analysis of chemical degradation within the system are illustrated.

Pope Marsh is a sub-alpine marsh at the outlet of the Upper Truckee River draining into Lake Tahoe. The marsh has been the focus of several recent studies because of its role in

Fig. 6 San Pablo Bay tidal marsh following channel enhancement

Fig. 7 San Pablo Bay tidal marsh during flooding prior to channel enhancement

limiting adverse nutrient and sediment loading of Lake Tahoe from the Upper Truckee watershed (Green 1998). Marsh integrity and function was compromised by development of a large condominium complex on part of the site and subsequent groundwater pumping. Pope Marsh overlies wave-dominated deltaic deposits in which sediments distant from the Lake are fluvial in origin while better-sorted wave-distributed sediments exist near the Lake. Adjacent to the Lake, these coarse-grained sand deposits are interconnected and result in rapid, preferential flow paths between the Lake and the Marsh. Whereas, fluvially-deposited sediments further upstream are very heterogeneous and subject to variable flows. Ecological communities of the northern portion of the Marsh near the Lake depend on adequate Lake water levels to maintain hydric conditions, while the southwest portion of the Marsh distant from the Lake has historically relied on groundwater for maintenance and is relatively insensitive to Lake water levels. These hydrogeologic conditions directly effect rates of peat development and water quality within the Marsh. Increased groundwater pumping and decreased surface water inflows have already reduced water quality within the Marsh, as well as the quantity and diversity of hydrophytic vegetation. Such changes have impaired Marsh functionality in terms of water quality enhancement of Truckee River waters flowing into the Lake.

Though in a coastal setting, a very similar linkage between wetland hydraulics, water quality and vegetation was observed in a tidal salt marsh on the north side of San Pablo Bay. Excessive fatality of the threatened harvest mouse and loss of pickleweed (Salicornia spp.) stand in the tidal marsh, prompted efforts to restore the wetlands (Grismer et al. 1998). Two open-channel systems were designed and installed in the fall of 1996 as a means of improving the tidal cycling of water in the marsh as well as the drainage of excess surface waters. In the approximately 200-hectare site, nearly 11.5 km of drainage channels were installed having a surface area of roughly 18 600 m^2. This drainage density was less than that of other natural sites due to the desire to maintain some flooding (approximately 35-40%) of the marsh and to encourage natural drainage channel development. In post-construction surveys (18 months after channel installation), channels constructed in healthy pickleweed stands were found to be stable, nearly rectangular in cross-section and had not filled with sediment, whereas channels lacking pickleweed growth adjacent to the channels had silted in substantially and taken on a broad V-shaped cross section. In addition, several "natural" channels, varying in cross-section from several cm wide and a few cm deep to approximately 1 m wide and 0.3 m deep, had formed in the flooded, relatively bare areas of the excavation spoils. The predominant difference in channel geomorphology was also related to tidal prism volumes and ebb tide outflow durations experienced by the channels. The primary factor affecting water quality within the marsh could be related to the large magnitude of the spring flood tide inflow velocities, period and sediment load. Though ebb tide outflow from the marsh began shortly after high tide levels were reached in the Bay, outflow continued for several hours after low tide in the Bay until sufficient flood tide elevation in the Bay was reached to reverse marsh channel flows.

Pre-channel construction air photos and satellite infrared surveys were used to characterize the site in terms of vegetation coverage. These images were digitized and calibrated with ground-truth frame surveys to establish that the pre-construction (October 1996) site conditions were 60% bare soil and 40% vegetation by area. By September 1997, a survey of the site found 63% vegetation, 23% bare soil and 14% channel excavation spoil, or inundated areas. The introduction of tidal flows to the wetlands combined with improved winter drainage appears to have encouraged significant emergence of the pickleweed (with no additional reseeding) as a result of the drainage. Some areas of the marsh, however, do not appear to be responding and some areas remain flooded as designed. In addition, natural extensions of some of the drainage channels have developed further enhancing the site conditions.

Surface water flow hydraulics also play an important role in constructed wetland management and performance. The Sacramento Regional County Sanitation District developed a 10-ha free surface constructed wetland system for tertiary treatment of municipal effluent such that remaining nitrogen, phosphorous and metals loading were reduced prior to eventual stream discharge (Nolte and Assoc 1997). Water balance studies indicated that on an annual basis rainfall accounted for 4%, evapotranspiration 2.5%, and surface outflow 68%, respectively, of the total inflow with the remainder presumably lost to seepage. Tracer studies indicated that the wetland cell detention times ranged from 10.4 to 12 days, or 10 to 20% larger than the theoretical plug-flow detention times. Non-uniformity in flow was not considered, nor the potential losses to groundwater in computing the detention times. Overall nitrogen removal efficiencies depended on season (due to changing water temperatures) resulting in average reduction of TKN concentrations from 14.5 mg/*l* to 5.8 mg/*l* in the summer and 14.5 mg/*l* to 12.7 mg/*l* in the winter. Average concentration-based removal rates for metals ranged from 34 to 89% with the bulk of the metals being adsorbed to the soil and sediments. Vegetation also played a significant role in accumulating metals through their adsorption in the "mat" and "peat" deposition layers on the soil. However, metal removal rates and efficiencies depended on the particular metal; silver, cadmium, copper, mercury and zinc were readily removed from the water column, while arsenic, chromium, nickel and lead showed little decrease in concentration with distance along the wetland cell. Some of this variability is due to the relatively low concentrations of all the metals, as well as due to the variability in ion valence conditions (e.g., As and Cr). Metal and BOD removal were modeled as first-order decay reactions with the modification of BOD removal to include a non-reducible, background concentration of 3.1 mg/L following the k-C* formulation proposed by Kadlec (1996). Shepherd (1998) and Shepherd et al. (1998) have proposed a rate-dependent decay constant that incorporates the variable detention times and slow breakdown of recalcitrant compounds in the wastewater. Improving the chemical uptake/removal models will require better characterization of the water flow paths in the system, potentially changing redox conditions and the nature of the particular compounds of interest.

In order to begin assessment of the link between constructed wetland treatment performance and subsurface flow hydraulic conditions, Grismer et al. (1997, 1998) and Shepherd (1998) and Shepherd et al. (1998) evaluated the hydraulic characteristics of two pilot-scale constructed wetland tanks designed to reclaim winery wastewater. Their studies included variations in water flowrates and chemical removal rates with distance and depth along tanks having established and no vegetation; in particular, they examined the lateral and longitudinal flow symmetry, residence times and dispersion coefficients at different depths and distances. Each wetland tank (6.10 m long by 2.44 m wide by 1.20 m deep) contained pea gravel to a depth of 0.95 m. The "old" tank had established bulrush and cattail vegetation and had been operating as a treatment system for over three years while the "new" tank was constructed in exactly the same manner as the "old" tank and had no vegetation. Two impulse-type bromide-tracer studies were conducted to derive the hydraulic data necessary to evaluate the residence time and dispersion coefficients for each tank. They found that flow patterns through the tanks were non-uniform laterally in the "new" tank and vertically in the "old" tank. In the "new" tank, more flow occurred along the south side as compared to the north side of the tank. However, this flow was more-or-less uniformly distributed across the depth of the tank. In contrast, the "old" tank had a stratified three-layer flow distribution in which there was no lateral asymmetry of flow in the tank. Most of the tracer mass (65%) moved with the "ideal" velocity at the center depth while at the 0.25 m depth, 20% of the tracer mass moved more quickly and the remaining 15% of the tracer mass at the base (0.95 m depth) flowed more slowly than the ideal velocity. The dispersion coefficient also depended on depth in the "old" tank in contrast to the "new" tank, while dispersion in the center layer was similar between the two tanks. However, the dispersion coefficient near the surface of the "old" tank was nearly four times greater than the overall average (due to an average mean velocity that is nearly twice that of the overall average), while at the bottom of the "old" tank the dispersion coefficient was only half of the overall average. Despite this difference in flow regimes, the apparent outflow tracer dispersion, or Peclet numbers were about the same for both the "old" and "new" tanks. The relatively large Pe number of 35, or Dispersion number of 0.03 for the tanks suggest that advective/dispersive processes dominate the tracer movement similar to that observed by several others for subsurface flow wetlands (Kadlec and Knight 1996; Table 9-5, p.255).

CLOSURE

Wetland ecosystems, whether managed or natural, are a complex set of multiple interrelated processes that affect the water quality and biological community of the wetland. It is apparent that complete description of the wetland hydrogeologic setting is critical to adequate characterization of the dominant flow processes affecting the wetland water quality and biological communities. In addition, attention must be given to determination of the actual water flowpaths through the wetland system in order to characterize changing water quality. Finally, chemical degradation/retention processes require additional investigations that are chemical species, or class dependent. It is hoped that with more detailed knowledge of many of the mechanistic processes of importance to wetland systems that better models can be developed for both wetland restoration/management, or constructed wetland design considerations.

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