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Nutrient Criteria Technical Guidance Manual

Wetlands

Full Guidance Document available in full at:
http://www.epa.gov/waterscience/criteria/nutrient/guidance/

Chapter 2 Overview of Wetland Science

2.1 INTRODUCTION

Wetlands exist at the interface between terrestrial and aquatic environments. They serve as sources, sinks, and transformers of materials. (Figure 2.1) Wetlands serve as sites for transformation of nutrients such as nitrogen (N) and phosphorus (P). Dissolved inorganic forms of N and P are assimilated by microorganisms and vegetation and incorporated into organic compounds. Nitrate in surface- and ground-water is reduced to gaseous forms of N (NO, N_2O, N_2) by microorganisms, a process known as denitrification, and returned to the atmosphere. Phosphorus undergoes a variety of chemical reactions with iron (Fe), aluminum (Al), and calcium (Ca) that depend on the pH of the soil, availability of sorption sites, redox potential, and other factors. These biogeochemical reactions are important in evaluating the nutrient condition (oligotrophic, mesotrophic, eutrophic) of the wetland and its susceptibility to nutrient enrichment.

Wetlands also generally are sinks for sediment, and wetlands that are connected to adjacent aquatic ecosystems (e.g., rivers, estuaries) may trap more sediment as compared to wetlands that lack such connectivity (Fryirs et.al., 2007; Mitsch and Gosselink, 2000; Dunne and Leopold 1978). Wetlands also may be sources of organic carbon (C) (Bouchard 2007; Raymond and Bauer 2001) and nitrogen (N) (Mitsch and Gosselink 2000; Mulholland and Kuenzler 1979) to aquatic ecosystems. Production of plant biomass (leaves, wood, and roots) from riparian, alluvial, and floodplain forests and from fringe wetlands such as tidal marshes and mangroves provide organic matter to support heterotrophic foodwebs of streams, rivers, estuaries, and nearshore waters (Mitsch and Gosselink 2000; Day et.al., 1989).

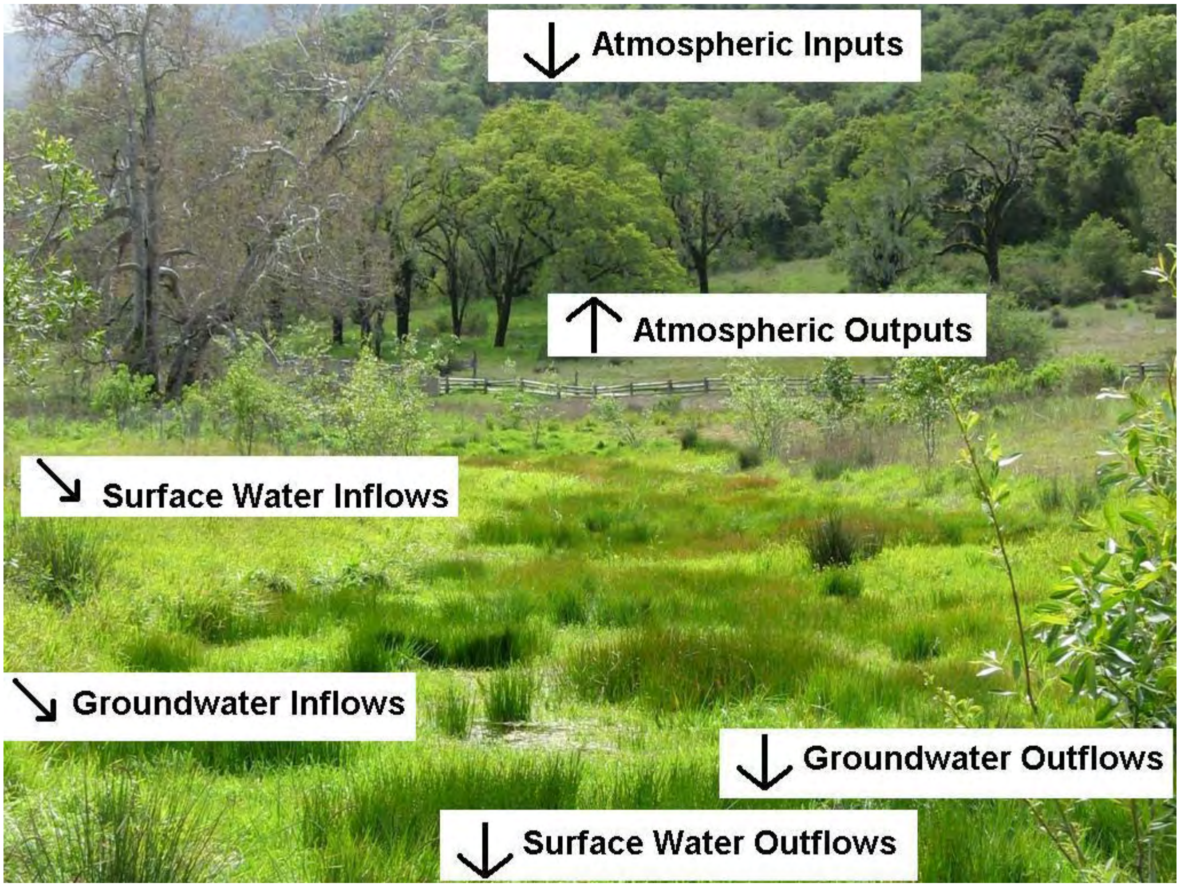

Figure 2.1. Schematic of nutrient transfer among potential system sources and sinks.

2.2 COMPONENTS OF WETLANDS

Wetlands are distinguished by three primary components: hydrology, soils, and vegetation. Wetland hydrology is the driving force that determines soil development, the assemblage of plants and animals that inhabit the site, and the type and intensity of biochemical processes. Wetland soils may be either organic or mineral, but share the characteristic that they are saturated or flooded at least some of the time during the growing season. Wetland vegetation consists of many species of algae, rooted plants that may be herbaceous and emergent, such as cattail (*Typha* sp.) and arrowhead (*Saggitaria* sp.), or submergent, such as pondweeds (*Potamogeton* sp.), or may be woody such as bald cypress (*Taxodium distichum*) and tupelo (*Nyssa aquatica*). Depending on the duration, depth, and frequency of inundation or saturation, wetland plants may be either obligate (i.e., species found almost exclusively in wetlands) or facultative (i.e., species found in wetlands but which also may be found in upland habitats). The discussion that follows provides an overview of wetland hydrology, soils, and vegetation, as well as aspects of biogeochemical cycling in these systems.

HYDROLOGY

Hydrology is characterized by water source, hydroperiod (depth, duration, and frequency of inundation or soil saturation), and hydrodynamics (direction and velocity of water movement). The hydrology of wetlands differs from that of terrestrial ecosystems in that wetlands are inundated or saturated long enough during the growing season to produce soils that are at least periodically deficient in oxygen. Wetlands differ from other aquatic ecosystems by their shallow depth of inundation that enables rooted vegetation to become established, in contrast to deep water aquatic ecosystems, where the depth and duration of inundation can be too great to support emergent vegetation. Anaerobic soils promote colonization by vegetation adapted to low concentrations of oxygen in the soil.

Wetlands primarily receive water from three sources: precipitation, surface flow, and groundwater (Figure 2.2). The relative proportion of these hydrologic inputs influences the plant communities that develop, the types of soils that form, and the predominant biogeochemical processes. Wetlands that receive mostly precipitation tend to be “closed” systems with little exchange of materials with adjacent terrestrial or aquatic ecosystems. Examples of precipitation-driven wetlands include “ombrotrophic” bogs and depressional wetlands such as cypress domes and vernal pools. Wetlands that receive water mostly from surface flow tend to be “open” systems with large exchanges of water and materials between the wetland and adjacent non-wetland ecosystems. Examples include floodplain forests and fringe wetlands such as lakeshore marshes, tidal marshes, and mangroves. Wetlands that receive primarily groundwater inputs tend to have more stable hydroperiods than precipitation- and surface water-driven wetlands, and, depending on the underlying bedrock or parent material, high concentrations of dissolved inorganic constituents such as calcium (Ca) and magnesium (Mg). Fen wetlands and seeps are examples of groundwater-fed wetlands.

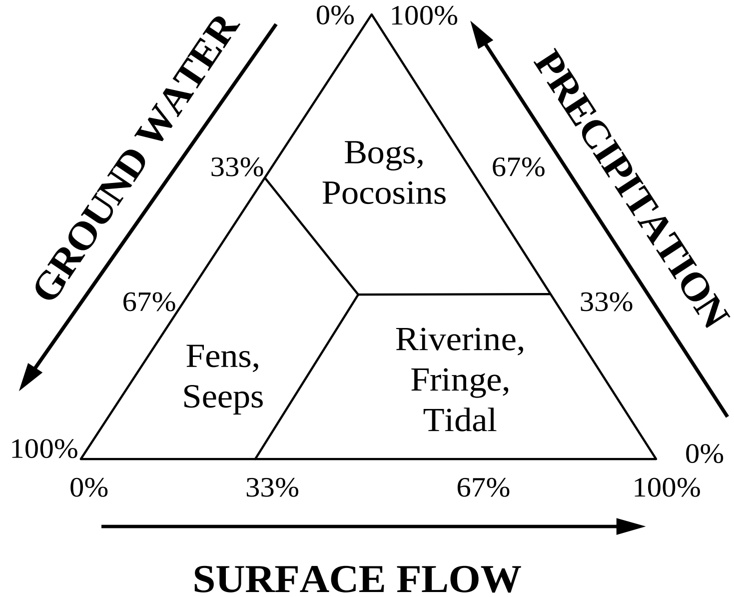

Figure 2. Relationship between water source and wetland vegetation. Modified from Brinson (1993).

Hydroperiod is highly variable depending on the type of wetland. Some wetlands that receive most of their water from precipitation (e.g., vernal pools) have very short duration hydroperiods. Wetlands that receive most of their water from surface flooding (e.g., floodplain swamps) often are flooded longer and to a greater depth than precipitation-driven wetlands. Fringe wetlands such as tidal marshes and mangroves are frequently flooded (up to twice daily) by astronomical tides but the duration of inundation is relatively short. In groundwater-fed wetlands, hydroperiod is more stable and water levels are relatively constant as compared to precipitation- and surface water-driven wetlands, because groundwater provides a less variable input of water throughout the year.

Hydrodynamics is especially important in the exchange of materials between wetlands and adjacent terrestrial and aquatic ecosystems. In fact, the role of wetlands as sources, sinks, and transformers of material depends, in large part, on hydrodynamics. For example, many wetlands are characterized by lateral flow of surface- or ground-water. Flow of water can be unidirectional or bidirectional. An example of a wetland with unidirectional flow is a floodplain forest where surface water spills over the river bank, travels through the floodplain, and re-enters the river channel some distance downstream. In fringe wetlands such as lakeshore marshes, tidal marshes, and mangroves, flow is bidirectional as wind-driven or astronomical tides transport water into, then out of the wetland. These wetlands have the ability to intercept sediment and dissolved inorganic and organic materials from adjacent systems as water passes through them. In precipitation-driven wetlands, flow may occur more in the vertical direction as rainfall percolates through the wetland soils to underlying aquifers or nearby streams. Wetlands with lateral surface flow may be important in maintaining water quality of adjacent aquatic systems by trapping sediment and other pollutants. Surface flow wetlands also may be an important source of organic C to aquatic ecosystems as detritus, particulate C, and dissolved organic C are transported out of the wetland into rivers and streams down gradient or to adjacent lakes, estuaries, and nearshore waters.

SOILS

Wetland soils, also known as hydric soils, are defined as "soils that formed under conditions of saturation, flooding, or ponding long enough during the growing season to develop anaerobic conditions in the upper part" (NRCS 1998). Anaerobic conditions result because the rate of oxygen diffusion through water is approximately 10,000 times less than in air. Wetland soils may be composed mostly of mineral constituents (sand, silt, clay) or they may contain large amounts of organic matter. Because anaerobic conditions slow or inhibit decomposition of organic matter, wetland soils typically contain more organic matter than terrestrial soils of the same region or climatic conditions. Under conditions of near continuous inundation or saturation, organic soils (histosols) may develop. Histosols are characterized by high organic matter content, 20-30% (12-18% organic C depending on clay content) with a thickness of at least 40 cm (USDA 1999). Because of their high organic matter content, histosols possess physical and chemical properties that are very different from mineral wetland soils. For example, organic soils generally have lower bulk densities, higher porosity, greater water holding capacity, lower nutrient availability, and greater cation exchange capacity than many mineral soils.

Mineral wetland soils, in addition to containing greater amounts of sand, silt, and clay than histosols, are distinguished by changes in soil color that occur when elements such as Fe and manganese (Mn) are reduced by microorganisms under anaerobic conditions. Reduction of Fe leads to the development of grey or "gleyed" soil color as oxidized forms of Fe (ferric Fe, Fe^{3+}) are converted to reduced forms (ferrous Fe, Fe^{+2}). In sandy soils, development of a dark-colored, organic-rich surface layer is used to distinguish hydric soil from non-hydric (terrestrial) soil. An organic-rich surface layer, indicative of periodic inundation or saturation, is not sufficiently thick (<40 cm) to qualify as a histosol, which forms under near-continuous inundation.

Wetland soils serve as sites for many biogeochemical transformations. They also provide long and short term storage of nutrients for wetland plants. Wetland soils are typically anaerobic within a few millimeters of the soil-water interface. Water column oxygen concentrations are often depressed due to the slow rate of oxygen diffusion through water. However, even when water column oxygen concentrations are supported by advective currents, high rates of oxygen consumption lead to the formation of a very thin oxidized layer at the soil-water interface. Similar oxidized layers can also be found surrounding roots of wetland plants. Many wetland plants are known to transport oxygen into the root zone, thus creating aerobic zones in predominantly anaerobic soil. The presence of these aerobic (oxidizing) zones within the reducing environment in saturated soils allows for the occurrence of oxidative and reductive transformations to occur in close proximity to each other. For example, ammonia is oxidized to nitrate within the aerobic zone surrounding plant roots in a process called nitrification. Nitrate then readily diffuses into adjacent anaerobic soil, where it is reduced to molecular nitrogen via denitrification or may be reduced to ammonium in certain conditions through dissimilatory nitrate reduction (Mitsch and Gosselink 2000; Ruckauf et al., 2004; Reddy and Delaune, 2005). The anaerobic environment hosts the transformations of N, P, sulfur (S), Fe, Mn, and C. Most of these transformations are microbially mediated. The oxidized soil surface layer also is important to the transport and translocation of transformed constituents, providing a barrier to translocation of some reduced constituents. These transformations will be discussed in more detail below in *Biogeochemical Cycling.*

VEGETATION

Wetland plants consist of macrophytes and microphytes. Macrophytes include free-floating, submersed, floating-leaved, and rooted emergent plants. Microphytes are algae that may be free floating or attached to macrophyte stems and other surfaces. Plants require oxygen to meet respiration demands for growth, metabolism, and reproduction. In macrophytes, much (about 50%) of the respiration occurs below ground in the roots. Wetland macrophytes, however, live in periodically to continuously-inundated and saturated soils and, therefore, use specialized adaptations to grow in anaerobic soils (Cronk and Fennessy 2001). Adaptations consist of morphological/anatomical adaptations that result in anoxia avoidance, and metabolic adaptations that result in true tolerance to anoxia. Morphological/anatomic adaptations include shallow roots systems, aerenchyma, buttressed trunks, pneumatophores (e.g., black mangrove (*Avicennia* *germinans*)), and lenticles on the stem. These adaptations facilitate oxygen transport from the shoots to the roots where most respiration occurs. Many wetland plants also possess metabolic adaptations, such as anaerobic pathways of respiration, that produce non-toxic metabolites such as malate to mitigate the adverse effects of oxygen deprivation, instead of toxic compounds like ethanol (Mendelssohn and Burdick 1988).

Species best adapted to anaerobic conditions are typically found in areas inundated for long periods, whereas species less tolerant of anaerobic conditions are found in areas where hydroperiod is shorter. For example, in southern forested wetlands, areas such as abandoned river channels (oxbows) are dominated by obligate species such as bald cypress (*Taxodium distichum*) and tupelo gum (*Nyssa aquatica*) (Wharton et.al., 1982). Areas inundated less frequently are dominated by hardwoods such as black gum (*Nyssa sylvatica*), green ash (*Fraxinus pennsylvanicus*), and red maple (*Acer rubrum*), and the highest, driest wetland areas are dominated by facultative species such as sweet gum (*Liquidambar styraciflua*) and sycamore (*Platanus occidentalis*) (Wharton et.al., 1982). Herbaceous-dominated wetlands also exhibit patterns of zonation controlled by hydroperiod (Mitsch and Gosselink 2000).

In estuarine wetlands such as salt- and brackish-water marshes and mangroves, salinity and sulfides also adversely affect growth and reproduction of vegetation (Webb and Mendelssohn 1996; Mitsch and Gosselink 2000). Inundation with seawater brings dissolved salts (NaCl) and sulfate. Salt creates an osmotic imbalance in vegetation, leading to desiccation of plant tissues. However, many plant species that live in estuarine wetlands possess adaptations to deal with salinity (Winchester et al., 1985; Whipple et al., 1981; Zheng et.al., 2004). These adaptations include salt exclusion at the root surface, salt secreting glands on leaves, schlerophyllous (thick, waxy) leaves, low transpiration rates, and other adaptations to reduce uptake of water and associated salt. Sulfate carried in by the tides undergoes sulfate reduction in anaerobic soils to produce hydrogen sulfide (H_2S) that, at high concentrations, is toxic to vegetation. At sub-lethal concentrations, H_2S inhibits nutrient uptake and impairs plant growth.

SOURCES OF NUTRIENTS

Point Sources

Point source discharges of nutrients to wetlands may come from municipal or industrial discharges, including stormwater runoff from municipalities or industries, or in some cases from large animal feeding operations. Nutrients from point source discharges may be controlled through the National Pollutant Discharge Elimination System (NPDES) permits, most of which are administered by States authorized to issue them. In general, point source discharges that are not stormwater related are fairly constant with respect to loadings.

Nonpoint Sources

Nonpoint sources of nutrients are commonly discontinuous and can be linked to seasonal agricultural activity or other irregularly occurring events such as silviculture, non-regulated construction, and storm events. Nonpoint nutrient pollution from agriculture is most commonly associated with row crop agriculture and livestock production that tend to be highly associated with rain events and seasonal land use activities. Nonpoint nutrient pollution from urban and suburban areas is most often associated with climatological events (rain, snow, and snowmelt), when pollutants are most likely to be transported to aquatic resources.

Urban and agricultural runoff is generally thought to be the largest source of nonpoint source pollution; however, growing evidence suggests that atmospheric deposition may have a significant influence on nutrient enrichment, particularly from nitrogen (Jaworski et.al., 1997). Gases released through fossil fuel combustion and agricultural practices are two major sources of atmospheric N that may be deposited in waterbodies (Carpenter et.al., 1998). Nitrogen and nitrogen compounds formed in the atmosphere return to the earth as acid rain or snow, gas, or dry particles. Atmospheric deposition, like other forms of pollution, may be determined at different scales of resolution. More information on national atmospheric deposition can be found at: http://www.arl.noaa.gov/research/programs/airmon.html and http://nadp.sws.uiuc.edu/. These national maps may provide the user with information about regional areas where atmospheric deposition, particularly of nitrogen, may be of concern. However, these maps are generally low resolution when considered at the local and site-specific scale and may not reflect areas of high local atmospheric deposition, such as local areas in a downwind plume from an animal feedlot operation.

Other nonpoint sources of nutrient pollution may include certain silviculture and mining operations; these activities generally constitute a smaller fraction of the national problem, but may be locally significant nutrient sources. Control of nonpoint source pollutants focuses on land management activities and regulation of pollutants released to the atmosphere (Carpenter et.al., 1998).

2.3 WETLAND NUTRIENT COMPONENTS

NUTRIENT BUDGETS

Wetland nutrient inputs mirror wetland hydrologic inputs (e.g., precipitation, surface water, and ground water), with additional loading associated with atmospheric dry deposition and nitrogen transformation (Figures 2.5 and 2.6). Total atmospheric deposition (wet and dry) may be the dominant input for precipitation-dominated wetlands, while surface- or ground-water inputs may dominate other wetland systems.

The total annual nutrient load (mg-nutrients/yr) into a wetland is the sum of the dissolved and particulate loads. The dissolved load (mg-nutrients/s) can be estimated by multiplying the instantaneous inflow (L/s) by the nutrient concentration (mg-nutrients/L). EPA recommends calculating the annual load by the summation of this function over the year—greater loads may be found during periods of increased flow and EPA recommends monitoring during these intervals. Where continuous data are unavailable, average flows and concentrations may be used if a bias factor (Cohn et al., 1989) is included to account for unmeasured loads during high flows. Particulate loads (mg-nutrients/yr) can be estimated using the product of suspended and bedload inputs (kg-sediments/yr) and the mass concentrations (mg-nutrients/kg-sediment).

Surface-water nutrient inputs are associated with flows from influent streams, as well as diffuse sources from overland flow through the littoral zone. Ground-water inputs can also be concentrated at points (e.g., springs), or diffuse (such as seeps). The influence of allochthonous sources is likely to be greatest in those zones closest to the source.

Because wetlands generally tend to be low-velocity, depositional environments, they often sequester sediments and their associated nutrients. These sediment inputs generally accumulate at or near the point of entry into the wetland, forming deltas or levees near tributaries, or along the shoreline for littoral inputs. Coarser fractions (e.g., gravels and sands) tend to settle first; the finer fractions (silts, clays, and organic matter) tend to settle further from the inlet point. Particulate input from ground-water sources can usually be neglected, while particulate inputs from atmospheric sources may be important if local or regional sources are present.

Wetland nutrient outputs again mirror hydrologic outputs (e.g., surface- and ground-water), and loads are again estimated as the product of the flow and the concentration of nutrients in the flow. While evaporation losses from wetlands may be significant, there are no nutrient losses associated with this loss. Instead, loss of nutrients to the atmosphere may occur as a result of ammonia volatilization, as well as N_2O losses from incomplete denitrification. Because sediment outputs from wetlands may be minor, nutrient exports by this mechanism may not be important.

Nutrient accumulation in wetlands occurs when nutrient inputs exceed outputs. Net nutrient loads can be estimated as the difference between these inputs and outputs. It is important, therefore, to have some estimate of net accumulation by taking the difference between upstream and downstream loads. Sampling ground-water nutrient concentrations in wells located upstream and downstream of the wetland can provide some sense of net nutrient sequestration, while sampling wetland nutrient inflows and outflows is needed for determining the additional sequestration for this pathway.

BIOGEOCHEMICAL CYCLING

Biogeochemical cycling of nutrients in wetlands is governed by physical, chemical, and biological processes in the soil and water column. Biogeochemical cycling of nutrients is not unique to wetlands, but the aerobic and anaerobic interface generally found in saturated soils of wetlands creates unique conditions that allow both aerobic and anaerobic processes to operate simultaneously. The hydrology and geomorphology of wetlands (Johnston et.al., 2001) influences biogeochemical processes and constituent transport and transformation within the systems (e.g., water-sediment exchange, plant uptake, and export of organic matter). Interrelationships among hydrology, biogeochemistry, and the response of wetland biota vary among wetland types (Mitsch and Gosselink, 2000; Reddy and Delaune, 2005).

Biogeochemical processes in the soil and water column are key drivers of several ecosystem functions associated with wetland values (e.g., water quality improvement through denitrification, long-term nutrient storage in the organic matter) (Figure 2.3). The hub for biogeochemistry is organic matter and its cycling in the soil and water column. Nutrients such as N, P, and S are primary components of soil organic matter, and cycling of these nutrients is always coupled to C cycling. Many processes occur within the carbon, nitrogen, phosphorus, and sulfur (C, N, P, or S) cycles; microbial communities mediate the rate and extent of these reactions in soil and the water column.

Aerobic-anaerobic interfaces are more common in wetlands than in upland landscapes and may occur at the soil-water interface, in the root zones of aquatic macrophytes, and at surfaces of detrital tissue and benthic periphyton mats. The juxtaposition of aerobic and anaerobic zones in wetlands supports a wide range of microbial populations and associated metabolic activities, with oxygen reduction occurring in the aerobic interface of the substrate, and reduction of alternate electron acceptors occurring in the anaerobic zone (D'Angelo and Reddy, 1994a or b). Under continuously saturated soil conditions, vertical layering of different metabolic activities can be present, with oxygen reduction occurring at and just below the soil-floodwater interface. Substantial aerobic decomposition of plant detritus occurs in the water column; however, the supply of oxygen may be insufficient to meet demands and drive certain microbial groups to utilize alternate electron acceptors (e.g., nitrate, oxidized forms of iron (Fe) and manganese (Mn), sulfate, and bicarbonate (HCO_3)).

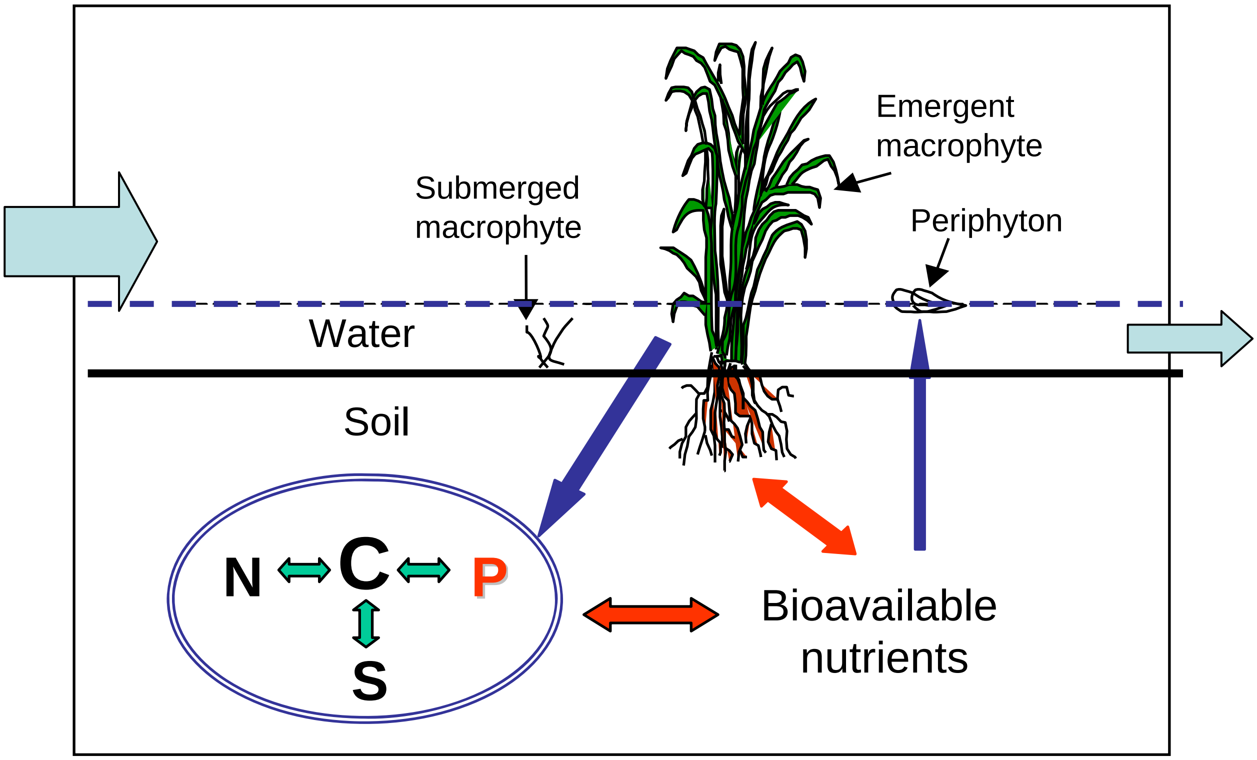

Figure 2.3 Schematic showing basic nutrient cycles in soil-water column of a wetland.

Soil drainage adds oxygen to the soil, while other inorganic electron acceptors may be added through hydraulic loading to the system. Draining wetland soil accelerates organic matter decomposition due to the introduction of oxygen deeper into the profile. In many wetlands, the influence of NO_3 and oxidized forms of Mn and Fe on organic matter decomposition is minimal. This is because the concentrations of these electron acceptors are usually low as a result of the fact that they have greater reduction potential than other alternate electron acceptors, so they generally are depleted rapidly from systems. Long-term sustainable microbial activity is then supported by electron acceptors of lower reduction potentials (sulfate and HCO_3). Methanogenesis is often viewed as the terminal step in anaerobic decomposition in freshwater wetlands, whereas sulfate reduction is viewed as the dominant process in coastal wetlands. However, both processes can function simultaneously in the same ecosystem and compete for available substrates (Capone and Kiene 1988).

A simple way to characterize wetlands for aerobic and anaerobic zones is to determine the oxidation-reduction potential or redox potential (Eh) of the soil-water column (Figure 2.4). Redox potential is expressed in units of millivolts (mV) and is measured using a voltmeter coupled to a platinum electrode and a reference electrode. Typically, wetland soils with Eh values >300 mV are considered aerobic and typical of drained soil conditions, while soils with Eh values <300 mV are considered anaerobic and are devoid of molecular oxygen (Figure 2.4).

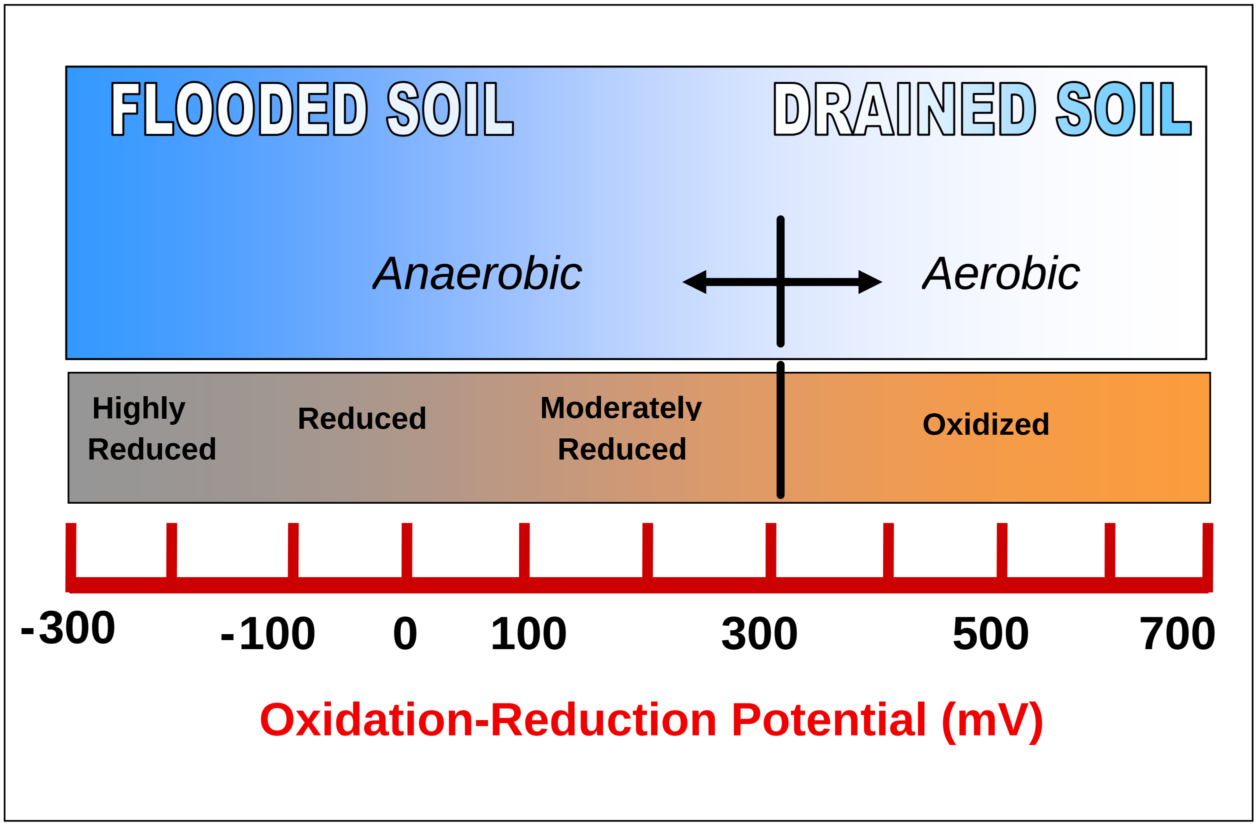

Figure 2.4. Range of redox potentials in wetland soils (Reddy and Delaune 2007).

Wetlands, as low-lying areas in the landscape, receive inputs from all hydrologically connected uplands. Many wetlands are open systems receiving inputs of carbon (C) and nutrients from upstream portions of the watershed that can include agricultural and urban areas.

Prolonged nutrient loading to wetlands can result in distinct gradients in water and soil. Mass loading and hydraulic retention time determine the degree and extent of nutrient enrichment. Continual nutrient loading to an oligotrophic wetland can result in a zone of high nutrient availability near the input, and low nutrient availability and possibly nutrient limiting conditions further from the input point. This enrichment effect can be seen in many freshwater wetlands, most notably in the sub-tropical Everglades where light is abundant and temperatures are high (Davis, 1991; Reddy et al., 1993; Craft and Richardson, 1993 a, b; DeBusk et al., 1994), and in some estuarine marshes (Morris and Bradley 1999). Between these two extremes, there can exist a gradient in quality and quantity of organic matter, nutrient accumulation, microbial and macrobiotic communities, composition, and biogeochemical cycles.

Compared to terrestrial ecosystems, most wetlands show an accumulation of organic matter, and therefore wetlands function as global sinks for carbon. Accumulation of organic C in wetlands is primarily a result of the balance of C fixation through photosynthesis and losses through decomposition. Rates of photosynthesis in wetlands are typically higher than in other ecosystems, and rates of decomposition are typically lower due to anaerobic conditions, hence organic matter tends to accumulate. In addition to maintaining proper functioning of wetlands, organic matter storage also plays an important role in regulating other ecosystems and the biosphere. For example, organic matter contains substantial quantities of N, P, and S; therefore, accumulation of organic matter in wetlands decreases transport of these nutrients to downstream aquatic systems.

NITROGEN (N):

Nitrogen enters wetlands in organic and inorganic forms, with the relative proportion of each depending on the input source. Organic forms are present in dissolved and particulate fractions, while inorganic N (NH_4-N, NO_3-N and NO_2-N) is present in dissolved fractions (Figure 2.5) or bound to suspended sediments (NH_4-N). Particulate fractions are removed through settling and burial, while the removal of dissolved forms is regulated by various biogeochemical reactions functioning in the soil and water column. Relative rates of these processes are affected by physico-chemical and biological characteristics of plants, algae, and microorganisms.

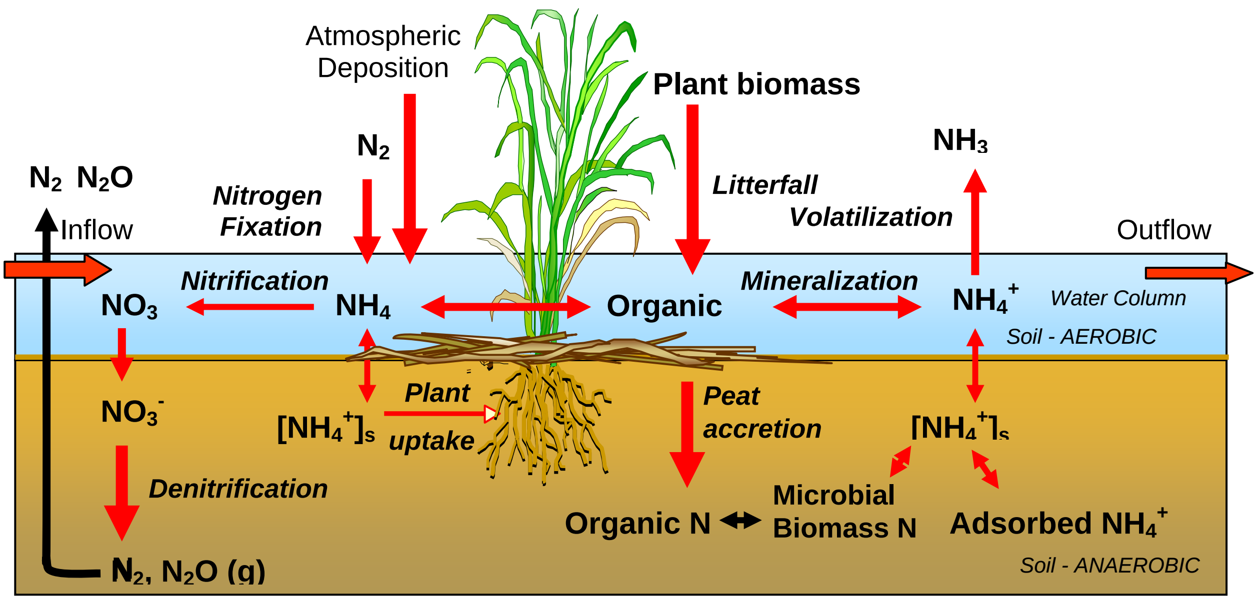

Figure 2.5. Schematic of the nitrogen cycle in wetlands.

Nitrogen reactions in wetlands effectively process inorganic N through nitrification and denitrification, ammonia volatilization, and plant uptake. These processes aid in lowering levels of inorganic N in the water column. A significant portion of dissolved organic N assimilated by plants is returned to the water column during breakdown of detrital tissue or soil organic matter, and the majority of this dissolved organic N is resistant to decomposition. Under these conditions, water leaving wetlands may contain elevated levels of N in organic form. Exchange of dissolved nitrogen species between soil and water column support several nitrogen reactions. For example, nitrification in the aerobic soil layer is supported by ammonium flux from the anaerobic soil layer. Similarly, denitrification in the anaerobic soil layer is supported by nitrate flux from the aerobic soil layer and water column. Relative rates of these reactions will, however, depend on the environmental conditions present in the soil and water column (Reddy and Delaune 2007).

PHOSPHORUS (P):

Phosphorus retention by wetlands is regulated by physical (sedimentation and entrainment), chemical (precipitation and flocculation), and biological mechanisms (uptake and release by vegetation, periphyton, and microorganisms). Phosphorus in the influent water is found in soluble and particulate fractions, with both fractions containing a certain proportion of inorganic and organic forms. Relative proportions of these pools depend on the input source. For example, municipal wastewater may contain a large proportion (>75%) as inorganic P in soluble forms, as compared to effluents from agricultural watersheds where a greater percentage of P loading may be in the particulate fraction.

Phosphorus forms that enter a wetland are grouped into: (i) dissolved inorganic P (DIP); (ii) dissolved organic P (DOP); (iii) particulate inorganic P (PIP); and, (iv) particulate organic P (POP) (Figure 2.6). The particulate and soluble organic fractions may be further separated into labile and refractory components. Dissolved inorganic P is generally bioavailable, whereas organic and particulate P forms generally must be transformed into inorganic forms before becoming bioavailable. Both biotic and abiotic mechanisms regulate relative pool sizes and transformations of P compounds within the water column and soil. Alterations in these fractions can occur during flow through wetlands and depend on the physical, chemical, and biological characteristics of the systems. Thus, both biotic and abiotic processes should be considered when evaluating P retention capacities of wetlands. Biotic processes include assimilation by vegetation, plankton, periphyton, and microorganisms. Abiotic processes include sedimentation, adsorption by soils, precipitation, and exchange processes between soil and the overlying water column (Reddy et.al., 1999, 2005; Reddy and Delaune, 2007). The processes affecting phosphorus exchange at the soil/sediment water interface include: (i) diffusion and advection due to wind-driven currents; (ii) diffusion and advection due to flow and bioturbation; (iii) processes within the water column (mineralization, sorption by particulate matter, and biotic uptake and release); (iv) diagenetic processes (mineralization, sorption, and precipitation dissolution) in bottom sediments; (v) redox conditions (O_2 content) at the soil/sediment-water interface; and, (vi) phosphorus flux from water column to soil mediated by evapotranspiration by vegetation.

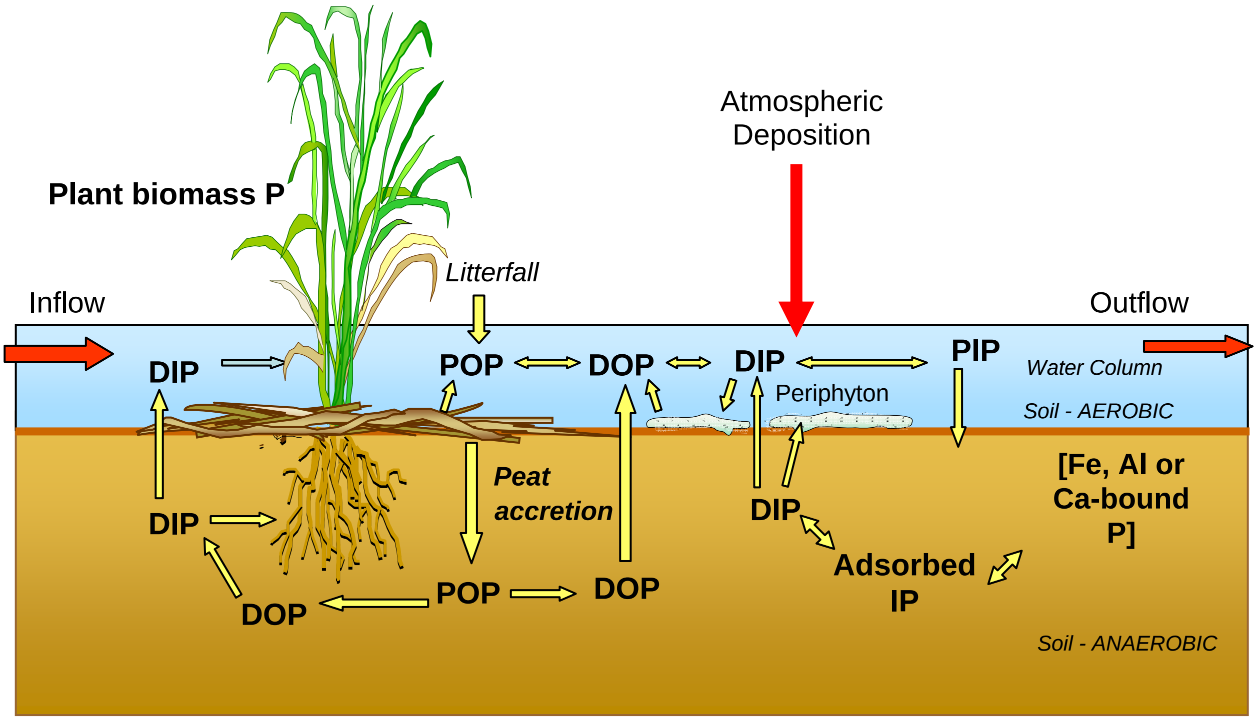

Figure 2.6. Schematic of the Phosphorous cycle in wetlands.

The key biogeochemical services provided by wetlands include nutrient transformation and removal by decreasing concentrations of nutrients and other contaminants, and sequestration of carbon and nutrients into stable pools (Kadlec and Knight 1996). The biogeochemical processes regulating water quality improvement are well established, and are made use of in treatment wetlands. Increased nutrient loading to oligotrophic wetlands results in increased primary productivity and nutrient enrichment. This resulting **eutrophication** can have both positive and negative impacts to the environment. Higher rates of primary productivity increase rates of organic matter accumulation, thus increasing **carbon sequestration**. However, eutrophication may lead to increased periodic and episodic export of DIP (Kadlec and Knight 1996; Reddy et.al., 1995, 1996, 2005; Reddy and Delaune 2007)).