Cumulative Impacts on Water Quality Functions of Wetlands

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ABSTRACT / The total effect of cumulative impacts on the water quality functions of wetlands cannot be predicted from the sum of the effects each individual impact would have by itself. The wetland is not a simple filter; it embodies chemical, physical, and biotic processes that can detain, transform, release, or produce a wide variety of substances. Because wetland water quality functions result from the operation of many individual, distinct, and quite dissimilar mechanisms, it is necessary to consider the nature of each individual process.

Sound knowledge of the various wetland processes is needed to make guided judgements about the probable effects of a given suite of impacts. Consideration of these processes suggests that many common wetland alterations probably do entail cumulative impact. In addition to traditional assessment methods, the wetland manager may need to obtain appropriate field measurements of water quality-related parameters at specific sites; such data can aid in predicting the effects of cumulative impact or assessing the results of past wetland management.

Wetlands fulfill a variety of roles in the landscape, from providing habitat to modulating water flow and influencing water quality. Water quality functions are a composite of many different biogeochemical processes, which act collectively to alter and usually improve the quality of surface waters.

Human impacts on wetlands may affect these water quality functions. Wetlands are frequently subject to multiple impacts over time and/or space; the effects of such multiple impacts may be simply additive, or the total effect may be more severe than the sum of the effects of the individual impacts alone. *Cumulative impact* as used here refers to multiple impacts whose effects on the wetland cannot be predicted by simply adding the effects of all the individual impacts. In a cumulative impact situation, assessment methodology must consider each impact in relation to others. This is akin to the concept of a nonlinear system as used in an engineering context.

The idea of cumulative impact is not new; it is implicit, for example, in the idea of a critical loading limit, beyond which a wetland's particular water quality functions will begin to fail.

To assess cumulative impact requires a detailed understanding of the individual water quality function of concern. Cumulative impact may not apply, for example, to purely chemical sorption of certain trace pollutants to a wetland soil; the process should be essentially linear over a reasonable range of pollutant loadings. It may be more likely if an impact alters the biota or hydrology of a wetland and thereby changes the underlying water quality mechanisms.

Traditional, intuitive notions of wetlands as filters are not adequate to assess the behavior of wetland water quality functions. We therefore begin this discussion of cumulative impact with an overview of wetland biogeochemical processes. We later consider several of the more common wetland alterations (impacts), the processes they are likely to affect, and the likelihood that cumulative impact is involved. The final section considers tools available to the wetland manager to assess the state of a wetland's water quality functions, and to help determine the likelihood that a cumulative impact situation exists. While no formal, generic, chemically appropriate assessment methodologies are available, several specific suggestions are made regarding the development of case-by-case wetland assessments and future research.

Overview of Wetland Water Quality Functions

An overview of the *in situ* processes that affect wetland and downstream water quality illustrates that wetlands are far from being simple sinks for pollutants. Wetlands often function as short- or long-term storage reservoirs, or even as sources for certain water quality constitutents. We will describe the major processes, recognizing that some overlap may occur between categories.

KEY WORDS: Wetlands; Water quality; Cumulative impact

Plant Nutrient Removal

Freshwater wetlands receive nitrogen (N) and phosphorus (P) from anthropogenic and natural sources. Anthropogenic sources include wastewater effluent discharge and runoff from urbanized and agricultural land. The major natural sources are nitrogen fixation, precipitation, runoff from vegetated watersheds, and groundwater inflow. The relative contribution of each type of input varies greatly among wetlands for both nitrogen and phosphorus (see Kelly and Harwell 1985 for examples). Nutrient loading rates are dictated by wetland topography, hydrology, setting (urban vs rural), and vegetation type.

Nutrients supplied to wetlands in effluent discharge and runoff are present in both soluble and particulate forms. Dissolved nitrogen is mostly introduced as nitrate (NO_3), ammonium (NH_4^+), or soluble organic forms. Dissolved phosphorus is received primarily as phosphate (PO_4^-) or soluble organic phosphorus (Kelly and Harwell 1985). Particulate species of both N and P include insoluble organics and sorbed ions.

Nitrogen can also enter wetlands via nitrogen fixation, the process whereby bacteria and algae associated with wetland soils, water, and plants transform atmospheric nitrogen into ammonium, which can be readily assimilated by plants. Nichols (1983) summarizes reported nitrogen-fixation rates in various types of wetlands.

The use of wetlands in wastewater treatment is well documented in the literature (see Whigham and Simpson 1976, Fetter and others 1978, Tilton and Kadlec 1979, Godfrey and others 1985).

There are three processes that immobilize or remove nitrogen from wetland waters: (1) accumulation by plants and microorganisms, (2) sedimentation, and (3) denitrification. Of these, only denitrification actually eliminates dissolved nitrogen from the system by releasing it to the atmosphere. The other two processes only immobilize nitrogen and phosphorus. The detained nutrients may be returned to solution when plants decompose, or they may be remobilized from sediments. In addition, sorbed and particulate nutrients may be transferred to adjacent waters when litter and sediments are flushed from the wetland. The interactions of these removal/release mechanisms determine the overall ability of a wetland to act as a sink for nutrients (Nichols 1983, Bayley and others 1985, Denton 1966, Howard-Williams 1985, Nixon and Lee 1985, Richardson 1985, Richardson and Marshall 1986).

Emergent vegetation obtains nitrogen and phosphorus primarily from the substrate; floating plants take up nutrients directly from the water, and rooted submergent plants can assimilate nutrients from both sediment and water (Nichols 1983). Aquatic plants in temperate regions actively absorb and utilize nutrients during the growing season, particularly in the early spring (Bernard and Solsky 1976). Accumulation ratios (nutrient concentration in plant tissues to nutrient concentration in surrounding waters) on the order of 10^3-10^4 have been observed for submersed aquatics (Peverley 1985).

Nutrients contained in green shoots during spring and summer may be translocated below ground for storage during the winter, then back above ground for use the following spring (Barko and Smart 1980). Some researchers have observed decreased nitrogen and phosphorus levels in aboveground tissues at the time of die-back relative to summer shoot nutrient content (Bernard and Solsky 1976). Others have not found evidence of significant translocation of nutrients for winter rhizome storage (Mason and Bryant 1975).

Internal cycling by aquatic plants reduces release of nutrients to wetland waters during die-back (Bernard and Solsky 1976). If, however, nutrients are not stored and recycled, senescence can lead to the return of large quantities of nitrogen and phosphorus to solution, through the leaching of nutrient-rich litter (Polunin 1984, Peverly 1985). Peverley (1985) found that "plants may contribute to nutrient movement from sediments back to the water by contributing a large fraction of total stream load of N, P, and C [carbon] in late summer and fall seasons as they senesce." Cycling of nutrients by wetland plants warrants further investigation, particularly seasonal transport of nutrients from shoots to rhizomes and nutrient release from decaying litter.

In the freshwater marsh studied by DeLaune and others (1986) "a large portion of the nitrogen incorporated in the vegetation [accumulates] mainly as organic nitrogen in accreted sediments." This illustrates that accumulation of organic sediment by wetland vegetation can facilitate relatively long-term nutrient detention.

The Mississippi delta marsh studied by DeLaune and others (1986) had a high peat accretion rate (0.75 cm/yr) and a high nitrogen accumulation rate (12 g N/m^2-yr). Accretion rates up to 2 mm/yr and nitrogen accumulation rates ranging from 0.10–4.7 g N/m^2-yr are typical of wetlands in colder climates (Nichols 1983). Thus, nutrient removal by organic soil development may be more important in warm, highly productive areas than in cooler regions. Phosphorus accumulation by this mechanism tends to be lower than nitrogen accumulation, but adsorption of phosphorus onto sediments may account for significant removal of this nutrient. This process is discussed below.

Chemical adsorption of phosphate (PO_4^{-3}) by hydrous iron (Fe) and aluminum (Al) oxides or silicate clays occurs by ligand exchange (Nichols 1983). Phosphate may also be physically sorbed by soils (Ryden 1978), or removed from solution by precipitation of insoluble Fe, Al, and calcium (Ca) phosphate (Nichols 1983). Sorbed ions are held out of solution by in-place sediment (Oshwald 1972). Phosphorus adsorption in wetlands is limited by the sorptive capacity of the suspended solids and sediments present (Edzwald 1977). The ability of a wetland to retain phosphorus through adsorption and precipitation is related to its capacity to trap mineral soils. Rates of sediment and associated nutrient accumulation vary greatly among wetland types. Johnston and others (1984) found that a seasonally flooded wetland that was accreting sediment at a rate of 2.0 kg/m^2-yr was accumulating phosphorus at a rate of 2.6 g P/m^2-yr. We note that such high rates of deposition could not continue indefinitely without seriously altering the wetland.

Organic soils tend to have lower capacity (Schneider and Erickson 1972, Childs and others 1977), but are still capable of retaining phosphorus (Bayley and others 1985). The phosphorus-detaining mechanisms do interact; for example, lowering of phosphorus concentration in wetland waters by plant uptake or dilution allows release of sorbed phosphorus back into solution (Nichols 1983).

In waterlogged, oxygen-depleted wetland soils, facultative anaerobic bacteria can use NO_3^- as the final electron acceptor in respiration. As a result, NO_3^- is reduced and released to the atmosphere as N_2O or N_2. Because nitrogen is thus lost from the wetland soil/water system, this process of denitrification can occur indefinitely without damaging a wetland. Rates of denitrification measured in wetlands of the northeastern United States averaged 4 g N/m^2-yr for submerged creek bottoms (Nixon and Lee 1985). In wetlands with an adequate supply of soil organic material, the rate of NO_3^- diffusion into the sediment may be the rate-limiting factor (Phillips and others 1978, Reddy and others 1978).

In summary, nitrogen and phosphorus exogenously supplied to wetlands can be removed from wetland waters in the short term by plant uptake, in the long term by peat and sediment accumulation, and permanently by denitrification. The percent nutrient removal by wetlands is generally high at low loading rates and low at greater loading rates (Nichols 1983). Furthermore, as evidenced by wetlands receiving effluent over several years, retentive capacity tends to decrease over time (Nichols 1983). The ability for wetlands to act as sinks for nitrogen and phosphorus is not limitless (Kadlec and Kadlec 1978). Introduction of high levels of nutrients over long periods of time can lead to deterioration of water quality in wetlands and export of nitrogen and phosphorus to adjacent water. More long-term studies on nutrient cycling in wetlands that have been receiving effluent for many years must be carried out.

Removal of Biological Oxygen Demand and Chemical Oxygen Demand

The biological oxygen demand (BOD) of water is a measure of oxygen required for the biogeochemical degradation of organic matter and the oxidation of inorganic matter such as sulfide. Chemical oxygen demand (COD) is a measure of the oxygen equivalent of the organic matter that can be oxidized by a powerful chemical oxidant. Since both parameters are positively correlated to gross organic content of waters, they are related to one another. Large amounts of both BOD and COD may be introduced to wetland waters along with sewage effluent, or in surface runoff containing organic matter. BOD and COD are also introduced by natural biotic processes in wetlands. If the BOD of a body of water is high relative to the rate of aeration, low dissolved oxygen levels or even anoxia will result. Therefore, removal of high levels of BOD and COD is one of the required functions of wetlands used in effluent treatment.

Wetlands decrease the BOD and COD of introduced waters through decomposition of organic material during aerobic bacterial respiration. DeJong (1976) studied wastewater purification in a rush pond and found BOD and COD removal was a function of residence time in the pond. He concluded that removal resulted from infiltration of wastewater into the sediment followed by decomposition by soil bacteria, as well as purification of through-flowing waters by microbes in the pond. The reed field under consideration accomplished nearly 100% BOD and COD removal in 16 days. In Tinicum marsh (Grant and Patrick 1970), significant reductions in BOD in the river flowing over the marsh were observed after 2–5 hr. This reduction occurred in 57% of the measurements, and the authors attribute inconsistency of removal to variability of flow and pollution load.

Both of these studies suggest that BOD removal is dependent on wetland hydrology. Wetlands with long residence times are probably best suited for BOD and COD reduction, as well as for other processes. While plant material is a source of BOD, the presence of wetland vegetation can also improve purifying capacity by trapping particulate organic matter and allowing sites of attachment for decomposing microorganisms.

Removal of Suspended Solids

Suspended solids may reach a wetland in runoff from the watershed, as particulate litterfall from vegetation, or via waterways that pass through the area (Kadlec and Kadlec 1978, Rumer 1977). Thus, particulate matter in wetland waters is significant for two reasons: the solids themselves may be detrimental, and/or they may serve to transport other materials that affect water quality (Boto and Patrick 1978). Turbidity caused by suspended solids attenuates light penetration, thereby decreasing photosynthesis and oxygen production. If the particulate substances are rich in organic matter, high BOD and COD may lower the dissolved oxygen (DO) level in wetland waters. In addition, the presence of suspended organic or inorganic sediments may decrease the rate of oxygen absorption and diffusion in streams (Alonso and others 1975), or cause excessive silting of riverbeds or reservoirs (Boto and Patrick 1978, Paulet and others 1972).

Toxic substances may be incorporated in or adsorbed on particulate matter entering wetlands. Deposition results in the removal of incorporated materials to the sediment where they may be taken up by plants, decomposed by microorganisms, or buried (Boto and Patrick 1978). Since sedimentation may allow particulate pollutants to be detained long enough for further breakdown to occur, it can be considered a primary wetland process leading to water quality improvement. Substances often associated with particulate matter in wetlands include nutrients, heavy metals, radionuclides, and xenobiotic organic pollutants. Sediment interactions of these substances are discussed below.

Wetlands facilitate sedimentation of suspended solids by slowing water velocities. Vegetation plays a major role in mechanical trapping and stabilization of sediments. Lee and others (1976) reported that stands of marsh vegetation were effective in reducing the turbidity of dredge disposal slurry. In the case of salt marshes, sediment entrapment helps to maintain the level of the marsh surface (Boto and Patrick 1978).

Flocculation of suspended solids at the freshwater–saltwater interface greatly enhances sedimentation in estuaries. Freshwater wetlands do not possess the high ionic strength necessary for flocculation. However, microbial colonization of particles can produce sticky surfaces leading to coagulation and settling of aggregates. Plant exudates may flocculate and remove fine particles (Lee and others 1976).

Kadlec and Kadlec (1978) summarize gross removal of suspended solids in several wetland ecosystems. Reported values vary from 97% removal to a 250% increase. These data indicate that wetlands may serve as either a source or sink for suspended solids under different conditions.

Removal of Bacteria and Viruses

Along with nutrients and organic matter, sewage effluent entering wetlands contains large quantities of bacteria, particularly coliforms and pathogens such as *Salmonella* and *Enterococci*. Because of their potential health hazard to man, the presence of these organisms in surface water is highly undesirable.

Wetlands are able to reduce counts of pathogens entering in effluents. In the reed pond studied by DeJong (1976), bacterial contamination was greatly reduced in the effluent even during times of peak load. Seidel (1970) observed reductions in *E. coli, Salmonella,* and *Enterococci* in sewage sludge treated by *Phragmites* beds in as little as 5 min.

Probably the most important mechanism for bacteria removal by wetlands is simply detention while natural die-back occurs. Many pathogenic microorganisms found in sewage effluent cannot survive for long periods of time outside of their host organisms. It is also possible that protozoa present in shallow waters actively feed on bacteria, thereby speeding up the process of die-back. In addition, root excretions of wetland plants can kill pathogenic bacteria in contaminated waters (Seidel 1970).

Metals (Including Metallic Radioisotopes)

One major source of metals to waters of many wetlands is the atmosphere, via dry fallout or precipitation (Lazrus and others 1970, Pakarinen and Tolonen 1976, Damman 1978, Glooschenko and Capobianco 1978, Olsen 1983). The metals of most concern are trace metals or heavy metals, including toxic transition elements. Patterns for waters throughout the United States suggest that trace metal pollution is often primarily anthropogenic in origin. Greater concentrations are generally found in areas of heavy manufacturing and mining (Lazrus and others 1970). Other sources include metal-contaminated wastewater and dredging spoil effluents. In some lakes, significant amounts of trace metals may enter wetland waters with overland and groundwater inflow. The contribution of each source will vary from wetland to wetland.

Wetlands are usually considered to act as sinks for metals, but retentive capacity varies greatly for different metals as well as for different wetland types (Giblin 1985, Nixon and Lee 1985). Giblin's summary of the passage of metals through various types of wetlands shows measured values ranging from 0% of lead passing through an English bog to 100% of zinc passing through a North Carolina salt marsh.

Metal may be removed from solution by adsorption onto particulate matter. Iron and manganese often exist in natural waters as colloidal hydrous oxides, and other trace metals may be adsorbed onto these colloids or onto clays or organic macromolecules (Singer 1977). Settling of particulate matter sequesters sorbed metals into the sediment. Metals may also be sorbed directly onto already immobile sediment. Most metals are sorbed more efficiently by organic than by mineral soils (Vestergaard 1979). Since wetland sediments are usually rich in organic matter, they may be better suited for sorption of metals than are upland soils with less organic content. Sorption of most metals is favored by high pH and oxidizing conditions (Benoit 1972, Gambrell and others 1977) and inhibited by the formation of soluble complexes with dissolved organic carbon (DOC) (Singer 1977). Some metal cations also appear to form organically bound complexes with peat; in such cases, sorption may be essentially nonreversible under ordinary conditions (Wieder and Lang 1986).

Precipitation of metal compounds, followed by settling, also transfers metals from the water to the sediment. Metals in solution may precipitate out as hydroxides, carbonates, phosphates, and other salts (Benoit 1972). The solubility of many metals in aqueous systems depends their redox state and pH of solution. Under aerobic conditions, for example, the ferrous ion [Fe(II)] is converted to ferric ion [Fe(III)], which subsequently precipitates as hydrous ion oxides at ordinary pH. It should be noted, however, that for some metals (e.g., uranium and plutonium) the more oxidized species are typically more soluble.

DOC can affect chemical precipitation of metals. For example, tannic acid complexation of Fe(II) increases the solubility of iron in oxic waters by slowing the oxidation reaction (Singer 1977). Dissolved organic material can also increase the solubility of iron by complexing Fe(III) and preventing its precipitation as $Fe(OH)_3$ (Christman 1967).

Waterlogged wetland sediments are typically depleted of oxygen by microbial respiration. Under anaerobic conditions sulfate reduction can occur. Precipitation of most transition metals occurs in the presence of excess sulfide because metal sulfides are generally quite insoluble. This process is less important in freshwater than in saltwater wetlands because of the generally lower sulfate concentration in freshwater (Giblin 1985). In waters that do not contain appreciable sulfide ion, onset of anoxia in bottom sediments can also lead to release of reduced metal ions, such as ferrous iron, back into solution.

Metals are taken up from water and sediment by wetland plants (Simmers and others 1981). Whether or not assimilation by vegetation leads to removal of metals depends on the export of detritus from that system. In wetlands that are not actively exporting detritus, metals incorporated in plant material may be accumulated in organic sediments through peat formation. Decomposing vegetation releases organic compounds that can chelate metals and remobilize them from sediments (Giblin 1985). Therefore, by contributing DOC, vegetation may also be responsible for the transfer of metals from wetland sediment to the water.

In the case of mercury, plants can have an additional role. Kozuchowski and Johnson (1978) studied gaseous emission of mercury by *Phragmites communis* growing at the edges of a mercury-contaminated lake. They found that there was a positive correlation between emission by plants and concentration of mercury in the substrate, and concluded that "vascular plants may play an important role in the removal of mercury from soils and sediments by injecting it directly into the atmosphere."

Neutralization of Acid Deposition

Acid deposition consists largely of dilute nitric and sulfuric acids in precipitation, as well as sulfate and nitrate aerosols and gases such as SO_2 and HNO_3 vapor. Atmospherically deposited ammonium also has the potential to acidify receiving ecosystems upon biological uptake of the ammonium. Both sulfur and nitrogen undergo biologically mediated redox transformations, and in the more reduced forms are only weakly acidic or are basic in nature.

Because wetlands are in large part reducing ecosystems, it has been speculated that they may lessen the effects of acid deposition by reducing nitrate and sulfate, resulting in net alkalinity production in the ecosystem. Hemond (1980, 1983) demonstrated the near-quantitative removal of both nitric and sulfuric acids atmospherically deposited into an ombrotrophic *Sphagnum* bog. This result has been demonstrated in bog ecosystems across North America by Gorham and others (1985) and in an experimentally acidified fen in northwest Ontario (Urban and Bayley 1986, Bayley and others 1987). Evidence for important acid neutralization effects in other wetlands is seen in the low sulfate concentrations during the growing season in waters draining from catchments containing substantial wetlands area (LaZerte and Dillon 1984, Eshleman 1985, Hemond and others 1987b, see also Gorham and others 1986). The reducing regions of lakes also appear to have significant alkalinity-generating potential due to sulfate reduction (Schindler and others 1980, Kuivila and Murray 1984).

Sites experiencing seasonal desaturation can release stored sulfur through reoxidation and solubilization (Bayley and others 1986). It has been suggested that wetlands containing small amounts of iron would be more likely to store sulfur as organic sulfur, or release sulfur to the atmosphere as reduced sulfur gases, than to store sulfur as precipitated sulfide. However, the chemistry of sulfur in wetland sediments is extremely complex, and little is actually known about the detailed pathways of sulfur transformation in freshwater wetland sediments.

Sulfur and nitrogen reduction in wetlands, at present levels of acid deposition, cannot appreciably affect the mineralization of peat by serving directly as electron acceptors; sulfate and nitrate input rates are small compared to even modest wetland primary production. Fertilization effects or more subtle effects, such as an influence of more sulfidic porewaters on wetland biota, cannot be ruled out.

Removal of Xenobiotic Organic Pollutants

Freshwater wetlands may detain and/or chemically degrade xenobiotic organic pollutants. The two processes can be linked, as when a slowly degraded pollutant is delayed in its passage through a wetland ecosystem sufficiently long to allow degradative processes to occur.

One mechanism for the detention of dissolved organic pollutants in wetlands is sorption onto sediments. Browman and Chesters (1977) summarize sorption and desorption mechanisms of organic substances in relation to a variety of substrates. Sorption of nonpolar organic compounds may be modeled as dissolution of the compounds into soil organic matter. The degree of sorption of dissolved organic compounds onto soils (an index of detention as well as possible bioconcentration; see Ogata and others 1984) can be modeled using the concept of the partition coefficient (K_d), which is the ratio of the compound's concentration in the soil to its aqueous concentration. K_d can be estimated from the relationship $K_d = K_{oc} \times f_{oc}$, where K_{oc} is approximated by the octanol–water partition coefficient (K_{ow}) of a compound (Schwarzenbach and Westall 1981), and f_{oc} is the fraction of organic carbon in the sediment. Measured values of K_{ow} are tabulated for many compounds, and may also be estimated from pollutant chemical structure (Lyman and others 1982). Large values of K_{ow} correspond to more highly hydrophobic (nonpolar) compounds. Since wetland soils can have an f_{oc} approaching 100%, wetland soil has a high potential to detain nonpolar organic pollutants.

Wetlands can also be effective sites of degradation of organic pollutants. The extent of purely chemical degradation varies widely from compound to compound. The commonly low pH of many wetlands may favor acid-catalyzed hydrolysis at the expense of base-catalyzed hydrolysis. Shallow wetlands offer opportunities for photodegradation of pollutants (Zafiriou and others 1984). The surface waters of wetlands may also be the site of degradation by aerobic bacteria and fungi. Numerous summaries of our present knowledge of aerobic microbial degradation pathways are available (Colwell and Saylor 1978, Ghisalba 1983, Leisinger 1983, Slater and Lovatt 1984).

A unique role for wetlands in degrading organic pollutants may prove to be associated with anaerobic pathways. Anaerobic degradation of organic chemicals has not been well-documented; however, several workers (Parr and Smith 1976, Sleat and Robinson 1983, Suflita and others 1983) have shown that some organic compounds that cannot be degraded in aerobic environments can be degraded in reducing environments. For example, degradation of halomethanes occurs in anaerobic cultures (Bouwer and others 1981). Dichlorobenzene can be degraded, evidently by denitrifying bacteria, under anaerobic conditions (Kuhn and others 1985). The complete mineralization of certain aromatic compounds to carbon dioxide and water by anaerobic bacteria has also been demonstrated (Kong and Sayler 1983). Although it is not clear in many instances what are the actual pathways of degradation (for example, reductive dehalogenation might occur enzymatically or through a reduced intermediate compound, such as sulfide, produced by the bacterial community), the anaerobic environments of wetlands may prove to have important biodegradative capabilities for some organic pollutants. Conceivably, combined aerobic/anaerobic degradations, analogous to the conversion of ammonia to nitrogen gas in wetlands (Patrick and Tusneem 1972), may operate for some organic pollutants. The role of wetlands in the degradation of organic pollutants is clearly worthy of further investigation.

Two particularly important groups of organic pollutants, pesticides and polycyclic aromatic hydrocarbons (PAHs), have received particular compound-specific study regarding their fate in the environment.

Pesticides. Pesticides are transported to wetland waters directly from the atmosphere during aerial application or following volatilization. These compounds may also reach wetlands in runoff from treated land, in which case they are associated with suspended soil particles. For reviews on the sources and fates of pesticides in aquatic environments, see Gould (1972), Faust (1977), and Pionke and Chesters (1973).

Polycyclic aromatic hydrocarbons. PAHs are supplied to inland wetlands in industrial and domestic wastewaters. Storm sewer runoff from road surfaces is a particularly important source of PAHs in sewage. Direct runoff from land may also contain high concentrations of PAHs. Finally, these compounds can enter wetland waters directly from the atmosphere in rain or dry fallout. For a thorough treatment of sources and fates of PAH in aquatic environments see Neff (1979).

Humic Substance Production

Many wetlands and their drainage waters are acidic. Clymo (1964) cites precipitation, sulfur-metabolizing bacteria, secretion of organic acid molecules by live *Sphagnum*, and cation exchange as possible sources of hydrogen ions to bogs. While several of these mechanisms may operate simultaneously, the most important acidity production mechanism in most wetlands is now recognized as the production of humic substances, a large fraction of which consist of organic acids (Hemond 1980, Gorham and others 1985, Urban and Bayley 1986).

Dissolved humic substances from natural freshwaters are surprisingly consistent from location to location given the great variability in their sources. A model for the acid/base properties of humic substances is proposed by Oliver and others (1983). Data from other researchers (Hemond 1980, Cronan and Aiken 1985, Eshleman and Hemond 1985, McKnight and others 1985) are generally consistent with this model. To assess acidity at pH values above 5.5, humic substances from natural waters in North America can be modeled as strong acid at a concentration of approximately 7 meq/g of organic carbon.

In addition to lowering the pH of natural waters, humic substances influence the chemical speciation of metals, particularly the heavy metals and aluminum (Environmental and Social Systems, Ltd. 1985). Humic substances affect light penetration in lakes, and present a slowly utilizable substrate for microbes as well as a measurable photochemical oxygen demand. Humic substances can reduce the toxicity of dissolved aluminum to fish (McAvoy and Male 1987; A. Screpetis, unpublished data). Some fraction of natural humic material may inhibit microbes (a common explanation for the preservation of materials in bogs). Finally, in waters that are chlorinated for human consumption, humic substances serve as a substrate for the production of halomethanes (McAvoy and Male 1987).

Wetlands are frequently regarded as major sources of humic substances to surface waters. High concentrations of humic material in upland soils are often cited as evidence that upland soils also contribute heavily to the DOC of natural waters. We hypothesize that wetlands are often the more important source of humic material for surface waters containing high DOC. High concentrations of humic substances in upland soils typically appear only in the upper layers. Because DOC levels in mineral soil groundwaters are typically low (adsorption and microbial mineralization account for the observed decreases in DOC with depth), surface waters derived from baseflow should in general contain only low levels of humic material.

The factors influencing humic substance production in wetlands, and the actual biochemical pathways whereby humic substances arise, are not well understood (Thurman 1985). It is therefore difficult to predict how wetland impacts will influence wetland DOC contributions to natural waters. This area deserves further research.

Organic Carbon Contributions to the Food Web

Primary productivity of wetlands has been studied by many investigators, and an excellent summary of the literature can be found in Gore (1983). Wetland primary productivity is often high, presumably because plants are not experiencing water stress, and leaching of nutrients from the substrate is minimal. Productivity (usually measured by harvesting techniques) frequently exceeds 1 kg dry mass/m^2−yr, although in some wetlands, such as bogs, where acidity and nutrient availability appear to inhibit plant growth, this figure is lower. Productivity in wetlands frequently occurs belowground, and may exceed measured aboveground productivity (Gore 1983). While much wetland primary production may be stored as peat or degraded by microbial metabolism, wetlands also support grazing-based and detrital food webs (e.g., Mitsch and Gosselink 1986). The carbon fluxes that enter food webs can be extremely significant from the perspective of wetlands as habitat.

Trace Gas Production

Because they are large reservoirs of biologically active reducing capacity in close proximity to the atmosphere, wetlands can contribute important trace gases to the atmosphere. Such gases include methane (an important "greenhouse" gas) (e.g., Harriss and others 1985), dimethylsulfide and other reduced sulfur gases (of importance to the global sulfur cycle), and nitrous oxide (another "greenhouse" gas which is also of significance to atmospheric ozone chemistry).

The trace gas role of wetlands is not yet well understood, and the measurement of atmospheric fluxes to and from wetlands is difficult, due to the low concentrations and reactive nature of many of the gases involved and the difficulty in making measurements without disturbing the gas exchange process itself (Hemond and others 1987a). Ultimately, however, wetland exchanges with the atmosphere must be understood in order to establish a credible basis for assessing cumulative impacts on wetlands at a biospheric level.

Typical Cumulative and Noncumulative Impacts on Wetlands

Several common wetland impacts are likely to be cumulative, such as impacts that alter wetland biota. Water quality processes associated with biota are typically nonlinear. Cumulative impact is less likely when the underlying wetland processes behave linearly, such as several sorptive processes. Especially when pollutants are in low concentrations, chemical interactions with the sorbing system are often independent and linear. The detention of heavy metals, metallic radionuclides, and nonpolar organic compounds at moderate concentrations are proposed examples.

In the following sections we consider several specific activities that may alter water quality functions by altering hydrologic patterns, vegetation, or sediment chemistry. Specific attention will be paid to impacts that appear likely to bring about cumulative effects.

Hydrologic Manipulation

Most wetland water quality functions are tied directly or indirectly to hydrology, and many wetland impacts directly or indirectly affect hydrology. For example, where pollutant-bearing waters percolate through wetland soils, maximum exchange occurs between water, soil, and roots, and maximum retention may result. Where water runs across the surface of a wetland, exchange of solutes with the wetland may be controlled by the kinetics of the sediment-water interface. Settling of particles may be controlled by water velocity. Any processes involving reduction or oxidation depends strongly on the degree of sediment water saturation. Indeed, in nearly every case, wetland water quality processes must be considered in relationship to the hydrology of the wetland.

Many nonlinear relationships appear in wetland hydrologic processes, including water velocity vs erosive capability, water level vs water flow (stage-discharge relationships), and water table height vs sediment aeration. Accordingly, significant impacts on wetland hydrology should in general be examined as potential cumulative impacts.

Alteration of Vegetation

Altering a wetland plant community can alter the functions of the vegetation, with resulting effects on water quality. Plants are the primary source of chemical reducing capacity in wetlands, are major storage reservoirs for nutrients, and are important hydrologic agents as well. Wetland macrophytes facilitate mechanical trapping of suspended sediments, and are capable of assimilating some metals and chemical compounds. Submersed macrophytes and photosynthetic microorganisms are responsible for oxygen production during photosynthesis, while root exudates from wetland plants serve to kill pathogenic bacteria and aid in flocculation of fine suspended particles. Thus, alteration of vegetation creates many possibilities to alter water quality-related processes. Vegetational processes are often very nonlinear, so cumulative effects are likely to be important.

Pollutant Inputs

Pollutants may enter a wetland via either point or nonpoint sources. A nonpoint source is diffuse, so localized effects of extremely high concentrations are unlikely. Classic examples of nonpoint sources are atmospheric loading and agricultural runoff. Many wetland treatment schemes, where effluent inputs are spread out by a diffuser or spray nozzles, fall into this category as well. Nonpoint loading of wetland ecosystems by nutrients, such as nitrogen or phosphorus, is likely to alter a wetland community through eutrophication. Different wetland ecosystems vary widely in their response to nutrient inputs. For example, many *Sphagnum* species are known for their rather specific chemical habitats, and may be lost from a wetland under excess nutrient loading. Because threshold effects are likely, eutrophication may create a cumulative impact situation, as may inputs of phytotoxic materials, such as herbicides. On the other hand, if a permanent sink process is active and the pollutant is not retained in the system, cumulative impact may not occur. Probable examples include denitrification and pathogen removal.

Inputs of pollutants that accumulate by chemical sorption in wetland sediments (e.g., metals and trace levels of many organics) represent a troublesome matter of definition in that they may eventually break through a wetland even though their behavior in the wetland is linear, and cumulative impact, by our definition, is not involved. The intuitive association of pollutant breakthrough with the idea of cumulative impact may or may not be appropriate. Breakthrough eventually occurs even when a single loading is made on a linear throughflow system such as a laboratory column. Sediment accretion, burial, resuspension, and irreversible sorption can modify the situation in a real wetland.

Atmospheric deposition is a nonpoint discharge as herein defined. Primary atmospheric pollutants of concern include acids and their precursors (Gorham and others 1984), heavy metals (Schell 1986), radionuclides, and certain atmospherically-borne anthropogenic organic pollutants such as PCBs and DDT (Rappaport and others 1985). In extreme cases, atmospheric pollutants can seriously affect the wetland process that influences water quality. Destruction of vegetation in the vicinity of large smelters is a dramatic example. Outside of such extreme cases, it is unclear whether atmospheric inputs are often large enough to seriously alter the fundamental water quality processes of wetlands.

Point sources are similar to nonpoint sources in many of their effects. In addition, localized regions of high concentration may be established near point discharges. A point discharge may behave as a nonpoint discharge far from the source, but a variety of localized phenomena may occur, such as acute toxicity and physical erosion of sediment. Cumulative impacts appear likely in such situations.

Filling

Filling is one of the most common and severe impacts on wetlands. Accomplished all at once or over a long period of time, the final result is the destruction of the filled area and its biogeochemical functions. While a buried layer of organic material representing a former wetland will unquestionably continue to influence the quality of subsurface waters, water flow and chemical conditions are altered so drastically that it is difficult to compare the water quality role of an intact wetland with that of buried organic soil. Even partial filling of a wetland is likely to produce cumulative effects through nonlinear effects on hydrologic conditions in the remaining wetland.

Peat Harvesting

Peat harvesting may be complete or partial. Complete peat harvesting results in the total elimination of wetland processes. Although a harvested peatland may regrow, the timescale of such regeneration is very long. Partial harvesting of peat is often accomplished in layers starting at the surface, which is particularly likely to destroy water quality functions associated with vegetation. Equally important, harvesting operations are generally associated with drainage of the peatland, a hydrologic impact having substantial and nonlinear effects on the ability of the system to influence water quality. Cumulative impacts seem likely in peat harvesting.

Agriculture and Forestry

Agriculture has dramatic impacts on the water quality role of a wetland ecosystem. Agriculture usually involves initial removal of native vegetation, hydrologic manipulation (typically but not always limited to drainage), tillage, and the application of fertilizers and pesticides (Willrich and Smith 1970, Johnston 1986). While fertilization alone can lead to increased levels of nutrients in receiving waters, it is the authors' opinion that the hydrologic alterations associated with agriculture have the most profound influence on wetland water quality functions. Draining followed by planting of row crops is probably the most severe agricultural impact; the predictable result is subsidence of the soil due both to decrease in water content (which increases mechanical stress on the soil particles) and oxidation (Levesque and others 1982, Mathur and others 1981).

Forestry involving the harvesting of native tree species is somewhat less disruptive. Hydrological disruption is usually less severe, although drainage is practiced in many wetland forestry operations. Removal of tree cover represents the sudden loss of a major nutrient uptake capacity as well as a substantial change in the water and heat balance at the surface. Transpiration, drawing water from deeper subsurface regions, is partially replaced by evaporation from exposed wetland surfaces. The microclimate and soil temperatures will be less moderated once a forest cover is removed, and summertime soil temperatures may be higher. The harvesting of trees is expected to reduce the nutrient retention capacity of a wetland in the short term, although harvesting of biomass on a regular basis could become significant to the nutrient budget of a wetland and permit somewhat more effective long-term nutrient removal. Most of the above effects are inherently nonlinearly related to the extent of impact and hence represent a cumulative impact situation.

Wildlife Management

Wildlife management practices generally entail vegetation management and hydrologic alteration. Widely used methods include water-level regulation, herbivore management, and fire. Other methods include seeding and planting, basin deepening, island building, vegetation cutting or herbicide treatment, and artificial nest building (Weller 1981).

Water-level regulation is used to increase habitat values by improving interspersing of cover and water (Kadlec 1962). This goal may be accomplished by partial or complete drawdown to encourage growth of emergent vegetation, followed by flooding to attract wildlife. Conversely, water levels may be raised to flood out overly dense vegetation. Drawdown is especially likely to have major effects on sediment chemistry, as discussed earlier.

During drawdown of a waterfowl impoundment, total nitrogen levels in sediments may increase due to microbial nitrification, and nutrients may be released to the water (Kadlec 1962). Winter drawdown of shallow weedy lakes to freeze roots and other propagules is widely practiced; water weed vegetation is thereby profoundly altered and water quality processes involving vegetation may be altered. The relationship between impact and effect is likely to be quite nonlinear and hence cumulative.

Water Treatment

The use of wetlands for wastewater treatment has been well treated by several authors, as presented earlier. Such a high level of impact has dramatic effects on all aspects of wetland function. A rationale for such use of a wetland is that it may help create a globally improved environment, even if the individual wetland is greatly modified.

A newer application is the use of wetlands to treat acid mine drainage. Several water quality processes, notably sulfate reduction and metal sorption, are particularly relevant to this wetland application. Much of the work to date is summarized by Wieder and Lang (1984), Burns (1984), and Brooks and others (1985).

Assessing Cumulative Impacts on Wetlands

There appears to be no clear-cut *technical* difference between assessing cumulative and noncumulative impacts on wetlands. The major difference appears to be administrative, and consists of assessing multiple proposed wetland impacts as a whole rather than one by one.

The following points thus apply to any assessment of wetland impact, cumulative or not. We emphasize that wetland impact assessment is still an evolving art, not yet a science, and in practice is hindered by both resource limitations and gaps in the state of knowledge of wetland ecosystem behavior.

There are two practical dimensions to assessing impacts on wetlands: (1) determining the condition of a wetland relative to expected norms or baselines, and (2) predicting the effects of future impacts. Both are essential and mutually reinforcing.

Baseline conditions for wetland water quality-related functions are derived from observations of intact wetland ecosystems. Baseline conditions must be established with appropriate consideration of the large variability, both systematic and seemingly random, between wetlands. The organization of existing data on wetland water quality functions, and establishment of baselines, has not in general been done, but is an area worthy of further effort.

Prediction of impacts on wetlands may be based on numerical simulation models, statistical models, checklists, rules of thumb, and/or the intuition and judgement of experts. The eventual development of adequate predictive models will do much to facilitate accurate, fair, and economical impact assessment. Full coverage of such models is outside the scope of the present discussion. Clearly, however, the appropriate water quality processes from the section "Overview of Wetland Water Quality Functions" must be represented in any useful predictive model.

In the following section we discuss several aspects of wetland assessment that are common to both the establishment of water quality baselines and the development, testing, and calibration of predictive models. These may best be regarded as tools to assist the wetland manager and, in the future, to be integrated into formal assessment methodologies.

Measurement of Water Quality

Appropriate measurements of wetland water quality can be valuable to a wetland manager. However, it is well to be aware of the difficulties inherent in present measurement techniques. Obtaining samples of wetland porewaters is usually problematic because clays and sapric peats can readily clog most sampling devices. Since wetland porewaters are often anoxic, substantial chemical changes may occur between sampling and analysis if water samples are not rigorously protected from the atmosphere. Oxidation can have profound effects on observed chemical composition. Iron may precipitate and sorb transition metals, while sulfide may oxidize and release metals. Humic substances interfere with the chemistry of many standard analyses; standard, widely accepted, and properly performed analyses often give erroneous values for many chemical species in wetlands (nitrate and sulfate are prime offenders in this regard). For these reasons, the wetland manager (as well as the wetland scientist) should critically consider the analytical protocols on which the data set are based. A compilation and critical evaluation of analytical methods would be of considerable value to both the wetland research and management communities.

The need for *comprehensive* water quality measurements should not be overlooked. Unless rather specific issues of limited scope are involved, it is recommended that each major inorganic species, organic nitrogen, organic phosphorus, and total dissolved organic material be measured. Where anthropogenic pollutants are of concern, their distribution in sediments and waters should be determined. For all parameters, temporal and spatial variability may need to be addressed. While downstream measurement may provide appropriate spatial averaging, the time-varying aspects of wetland functions often call for observations of water quality over time. The correlations of water quality with biotic activity, season, and hydrologic activity may be important if the underlying processes are to be understood sufficiently to allow predictions to be made.

Physical Parameters Related to Water Quality Function

Physical parameters may be used to assess certain functions of a wetland. Of special interest are parameters such as channel structure and microtopography, which govern the movement of water within a wetland and the degree of contact between moving waters and the vegetation and sediment. The geometry of hydraulic structures is likely to have major effects on wetland water quality function, since these structures influence the seasonal pattern of water level and hence influence both redox conditions within the wetland sediment and long-term biotic vegetational patterns.

Pollutant Loadings and Budgets

Measured or calculated pollutant loading values are valuable to the wetland manager as they may indicate how threatened a wetland is by toxicity or eutrophication. Some loading values may be obtained fairly easily, e.g., many industrial or agricultural fluxes. Some sources, such as urban runoff, vary dramatically on an episodic basis. Such fluxes of pollutants may still be measured at reasonable cost if the loading episodes are relatively few and predictable.

Measurements of pollutant loading are particularly valuable when combined with values of pollutant export; the result is a pollutant budget for the wetland. Although costly to obtain in most situations, pollutant and nutrient budgets provide a direct measure of the degree to which wetland water quality functions are operating. Such budgets are essential to basic research on water quality functions and wetland modeling.

Biological Indicators

Certain biotic indicators provide integrated measures of the degree to which a wetland has been disturbed; such information may yield insights about the status of the water quality functions. For example, dense stands of *Phragmites communis* are often found in disturbed Northeastern wetlands. Existence of various species of moss may serve as pH or water-level indicators (Jeglum 1971). Certain diatom species are believed to be reliable pH indicators. Henbry and Cairns (1984) found that protozoan colonization rates serve as a reliable indicator of the trophic status of wetland lakes. Others have suggested the use of small-mammal or bird communities to monitor the ecological conditions of wetlands (Reicholf 1976, Steele and others 1984). It might prove possible to develop a set of biological indicators characteristic of wetlands suffering from certain impairments of their water quality functions, such as loss of nutrient retention ability, accumulations of excessive levels of heavy metals, or high levels of certain organic contaminants. We are aware of no ongoing systematic work on biological indicators in this role, but suggest that this may be an important avenue of future research.

Tissue Analysis

Since wetland plants and animals can assimilate chemicals from their environment, their tissues may accumulate contaminants found in the surrounding water and sediment (Simmers and others 1981, Larsson 1984, Aulio 1985, Gerloff and Krumbholz 1966). Tissue concentrations of substances such as heavy metals, radionuclides, and pesticide residues indicate the presence of these compounds in the substrate. Tissue concentrations of nutrients, however, may be difficult to interpret. For example, Bayley and others (1985) found that phosphorus content of vegetation in freshwater marsh plots varied as much with changing hydrology as with nutrient loading. More research is needed on tissue analysis as a tool for wetland assessment.

Sediment Analysis

Many workers have measured inventories of toxic compounds and other pollutants in the sediments of natural systems, including wetlands (e.g., Pakarinen and Tolonen 1976, Damman 1978, Glooschenko and Capobianco 1978). In principle, sediment analysis offers several attractive advantages for assessing the status of a wetland. The sediment inventory may best reflect the integrated history of the wetland. This is particularly useful where the manager must assess previous impacts on a particular site. Many pollutants are more concentrated in sediments than in water, thus simplifying the problem of making accurate measurements. More concentrated samples help minimize measurement artifacts and simplify analysis (although some measurements are more difficult to perform on solid samples). Finally, sediments are typically codeposited with biogeochemicals or biotic remains that serve as markers or geochronometers. Using lead-210, carbon-14, fallout radionuclides, charcoal strata, or microfossils, it may be possible to estimate directly not only the present burden of certain pollutants stored within a wetland's sediment, but also the historical role the wetland has had in removing these pollutants from the water (Schell 1986).

Compounds that are degraded by the wetland ecosystem are by their nature less likely to be found in the sediments. This is not a serious limitation, since such compounds are less likely to be associated with cumulative effects on the wetland water quality functions. On the other hand, recalcitrant organic pollutants and toxic metals will be clearly seen in the sediment profile. It may in principle be possible to estimate the degree to which degradation of wetland functions is associated with observed pollutant profiles.

Sediment analysis is potentially a very valuable tool for the wetland manager, and is a fruitful area for future research.

Conclusion

Wetlands influence water quality through a variety of biogeochemical processes. To date, assessment of impacts on these processes is hindered by both limitations in scientific knowledge and by the dearth of systematic, accurate, and practical methodologies to apply the science base to management problems. Several potentially useful measurements have been briefly outlined in this article. These should be compared for their overall utility to the wetland manager. Such comparisons would lead to better methods for assessing the water quality effects of impacts to individual wetlands.

Particularly in a cumulative impact context, it is necessary to study the interaction of water quality processes that occur in a wetland ecosystem. None of the water quality functions operate in isolation, and actual measurement of multiple processes at the same site will be required for a thorough understanding of the overall water quality roles of wetlands and the development of assessment techniques. Many-variable studies will require collaboration between researchers with specialities from several disciplines.

Finally, field studies of specific water quality functions over very long periods are needed to examine the long-term effects of wetland impacts. The changes in wetland processes that take place on the scale of years, decades, and longer are not adequately understood. The ambitious and time-consuming studies required would be facilitated by open cooperation between researchers, wetland managers, and funding agencies. The synthesis of information already available may also shed light on long-term impacts.

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Literature Cited

Aulio, K. 1985. Metal accumulation capacity of five species of *Sphagnum* moss. *Bulletin of Environmental Contamination and Toxicology* 35:439–442.

Banko, J. W., and R. M. Smart. 1980. Mobilization of sediment phosphorus by submersed freshwater macrophytes. *Freshwater Biology* 10:229–236.

Bayley, S. E., R. S. Behr, and C. A. Kelly. 1986. Retention and release of S from a freshwater wetland. *Water, Air, and Soil Pollution* 31:101–114.

Bayley, S. E., D. H. Vitt, R. W. Newbury, K. G. Beaty, R. S. Behr, and C. Miller. 1987. Experimental acidification of a Sphagnum dominated peatland: First year results. *Canadian Journal of Fisheries and Aquatic Sciences* 44:194–205.

Bayley, S. E., J. Zoltek Jr., A. J. Hermann, T. J. Dolan, and L. Tortora. 1985. Experimental manipulation of nutrients and water in a freshwater marsh: Effects on biomass, decomposition, and nutrient accumulation. *Limnology and Oceanography* 30(3):500–512.

Benoit, R. J. 1972. Washout processes in freshwater systems. Pages 45–61 *in* Proceedings of the Fifth Symposium on Water Resources Research. Ohio State University, Columbus, Ohio.

Bernard, J. M., and B. A. Solsky. 1976. Nutrient cycling in a *Carex lacustrus* wetland. *Canadian Journal of Botany* 55:630–638.

Boto, K. G., and W. H. Patrick, Jr. 1978. Role of wetlands in removal of suspended sediments. Pages 479–489 *in* P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. American Water Resources Association, Minneapolis, Minnesota.

Bouwer, E. J., B. E. Rittman, and P. L. McCarty. 1981. Anaerobic degradation of halogenated 1- and 2-carbon organic compounds. *Environmental Science and Technology* 15:596–599.

Brooks, R. P., D. E. Samuel, and J. B. Hill (eds.). 1985. Wetlands and water management on mined lands. Conference proceedings, 23–24 October. Pennsylvania State University, State College, Pennsylvania, 393 pp.

Browman, M. J., and G. Chesters. 1977. The solid-water interface: Transfer of organic pollutants across the solid-water interface. Pages 49–106 *in* I. H. Suffet (ed.), Fate of pollutants in the air and water environments, vol. I. John Wiley and Sons, New York.

Burns, J. E. (ed.). 1984. Treatment of mine drainage by wetlands. Contribution 264, Department of Biology, Pennsylvania State University, State College, Pennsylvania, 74 pp.

Childs, C. W., P. L. Searle, and N. Wells. 1977. Infiltration through soil as a tertiary treatment of sewage effluent. *New Zealand Journal of Science* 20:433–437.

Christman, R. F. 1967. The chemistry of rivers and lakes: The nature and properties of natural product organics and their role in metal ion transport. *Environmental Science and Technology* 1:302–306.

Clymo, R. S. 1964. The origin of acidity in *Sphagnum* bogs. *The Bryologist* 67:427–431.

Colwell, R. R., and G. S. Saylor. 1978. Microbial degradation of industrial chemicals. Pages 111–134 *in* R. Mitchell (ed.), Water pollution microbiology. John Wiley and Sons, New York.

Cronan, C. S., and G. R. Aiken. 1985. Chemistry and transport of soluble humic substances in forested watersheds of the Adirondack Park of New York. *Geochimica et Cosmochimica Acta* 49:1697–1705.

Damman, A. W. H. 1978. Distribution and movement of elements in ombrotrophic peat bogs. *Oikos* 30:480–495.

DeJong, J. 1976. The purification of wastewater with the aid of rush or reed ponds. Pages 133–139 *in* J. Tourbeir, and R. W. Pierson, Jr. (eds.), Biological Control of Water Pollution. University of Pennsylvania Press, Philadelphia, Pennsylvania.

DeLaune, R. D., C. J. Smith, and M. N. Sarafyan. 1986. Nitrogen cycling in a freshwater marsh of *Panicum hemitomon* on the deltaic plain of the Mississippi River. *Journal of Ecology* 74:249–256.

Denton, J. B. 1966. Relationships between the chemical composition of aquatic plants and water quality. M.S. thesis. Auburn University, Auburn, Alabama.

Edzwald, J. K. 1977. Phosphorus in aquatic systems: The role of the sediments. Pages 183–214 *in* I. H. Suffet (ed.), Fate of pollutants in the air and water environment, part I. John Wiley and Sons, New York.

Environmental and Social Systems, Ltd. (ESSA). 1985. The contribution of organic acidity to the acidification of surface waters and effects on aquatic biota: A structured review of existing knowledge. Environment Canada and Department of Fisheries and Oceans, Toronto, Ontario, Canada.

Eshleman, K. N. 1985. Hydrochemical response of a New England watershed to acid deposition. Ph.D. thesis, Massachusetts Institute of Technology, Cambridge, Massachusetts.

Eshleman, K. N., and H. F. Hemond. 1985. The role of organic acids in the acid–base status of surface waters at Bickford watershed, Massachusetts. *Water Resources Research* 21:1503–1510.

Faust, S. D. 1977. Chemical mechanisms affecting the fate of organic pollutants in natural aquatic environments. Pages 317–365 *in* I. H. Suffet (ed.), Fate of pollutants in the air and water environments, vol. II. John Wiley and Sons, New York.

Fetter, C. W., Jr., W. E. Sloey, and F. L. Spangler. 1978. Use of a natural marsh for wastewater polishing. *Journal of the Water Pollution Control Federation* 50:290–307.

Gambrell, R. P., R. A. Khalid, M. G. Verloo, and W. H. Patrick, Jr. 1977. Transformation of heavy metals and plant nutrients in dredged sediments as affected by oxidation–reduction potential and pH, vol. II. Materials and methods/results and discussion. Contract Report D-77-4. Dredged Material Research Program, Waterways Experiment Station, U.S. Army Corps of Engineers, Vicksburg, Mississippi.

Gerloff, G. C., and P. H. Krumbholz. 1966. Tissue analysis as a measure of nutrient availability for the growth of angiosperm aquatic plants. *Limnology and Oceanography* 11:529–537.

Ghisalba, O. 1983. Chemical wastes and their biodegradation: An overview. *Experimentia* 39:1247–1257.

Giblin, A. E. 1985. Comparison of processing of elements by ecosystems. II. Metals. Pages 158–179 *in* P. J. Godfrey, E. R. Kaynor, S. Pelczarski and J. Benforado (eds.), Ecological considerations in wetland treatment of municipal wastewaters. Van Nostrand Reinhold, New York.

Glooschenko, W., and J. Coppobianco. 1978. Metal content of *Sphagnum* mosses from two northern Canadian ecosystems. *Water, Air, and Soil Pollution* 10:215–220.

Godfrey, P. J., E. R. Kaynor, S. Pelczarski, and J. Benforado. 1985. Ecological considerations in wetland treatment of municipal wastewater. Van Nostrand Reinhold, New York, 474 pp.

Gore, A. J. P. (ed.). 1983. Ecosystems of the world. Vol. 4A, Mires: Swamp, Bog, Fen, and Moor. Elsevier, Amsterdam, 440 pp.

Gorham, E., S. E. Bayley, and D. W. Schindler. 1984. Ecological effects of acid deposition upon peatlands: A neglected field in acid rain research. *Canadian Journal of Fisheries and Aquatic Sciences* 41:1256–1268.

Gorham, E., S. J. Eisenreich, J. Ford, and M. V. Santleman. 1985. The chemistry of bog waters. Pages 339–363 in W. Stumm (ed.), Chemical Processes in Lakes. John Wiley and Sons, New York.

Gorham, E., J. K. Underwood, F. B. Martin, and J. Gordon Ogden III. 1986. Natural and anthropogenic causes of lake acidification in Nova Scotia. *Nature* 324(6096):451–453.

Gould, R. F. 1972. Fate of organic pesticides in aquatic environment. *Advances in Chemistry Series,* 111. American Chemical Society, Washington, D.C., 280 pp.

Grant, R. R., and R. Patrick. 1970. Tinicum Marsh as a water purifier. Pages 105–123 *in* Two studies of Tinicum Marsh. The Conservation Foundation, Washington, D.C.

Harriss, R. C., B. Gorham, D. I. Sebacher, K. B. Bartlett, and P. A. Flebbe. 1985. Methane flux from northern peatlands. *Nature* 315:652–653.

Hemond, H. F. 1980. Biogeochemistry of Thoreau's Bog, Concord, Massachusetts. *Ecological Monographs* 50:507–526.

Hemond, H. F. 1983. The nitrogen budget of Thoreau's Bog. *Ecology* 64:99–109.

Hemond, H. F., T. P. Army, W. K. Nuttle, and D. G. Chen. 1987a. Element cycling in wetlands: Interactions with physical mass transport. Pages 519–537 *in* R. Hites and S. Eisenreich (eds.), Sources and fates of aquatic pollutants. American Chemical Society, Washington, D.C.

Hemond, H. F., K. O'Connor, and B. Gordesky. 1987b. Surface hydrology and wetland roles in acid deposition on the Bickford Pond watershed. Report to the Massachusetts Division of Water Pollution Control, Westborough, Massachusetts, 71 pp.

Henbry, M. S., and J. Cairns, Jr. 1984. Protozoan colonization rates and trophic status of some freshwater wetland lakes. *Journal of Protozoology* 31:456–467.

Howard-Williams, C. 1985. Cycling and retention of nitrogen and phosphorus in wetlands: A theoretical and applied perspective. *Freshwater Biology* 15:391–431.

Johnston, C. A., G. D. Bubenzer, G. B. Lee, F. W. Madison, and J. R. McHenry. 1984. Nutrient trapping by sediment deposition in a seasonally flooded lakeside wetland. *Journal of Environmental Quality* 13:283–290.

Jeglum, J. K. 1971. Plant indicators of pH and water level in peatlands at Candle Lake, Saskatchewan. *Canadian Journal of Botany* 49:1661–1676.

Kadlec, J. A. 1962. The effects of drawdown on waterfowl impoundment. *Ecology* 43:267–281.

Kadlec, R. H., and J. A. Kadlec. 1978. Wetlands and water quality. Pages 436–456 *in* P. E. Greeson, J. R. Clark, and J. E. Clark (eds.), Wetland functions and values: The state of our understanding. American Water Resources Association, Minneapolis, Minnesota.

Kelly, J. R., and M. A. Harwell. 1985. Comparisons of processing of elements by ecosystems. I. Nutrients. Pages 137–157 *in* P. J. Godfrey, E. R. Kaynor, S. Pelczarski, and J. Benforado (eds.), Ecological considerations in wetlands treatment of municipal wastewater. Van Nostrand Reinhold, New York.

Kong, H. L., and G. S. Sayler. 1983. Degradation and total mineralization of monohalogenated biphenyls in natural sediment and mixed bacterial culture. *Environmental Microbiology* 46:666–672.

Kozuchowski, J., and D. L. Johnson. 1978. Gaseous emissions of mercury from an aquatic vascular plant. *Nature* 274:468–469.

Kuhn, E. P., P. J. Colberg, J. L. Schnoor, O. Wanner, A. J. B. Zehnder, and R. P. Schwarzenbach. 1985. Microbial transformations of substituted benzenes during infiltration of river water to groundwater: Laboratory column studies. *Environmental Science and Technology* 19:961–968.

Kuivila, K. M., and J. W. Murray. 1984. Organic matter diagenesis in freshwater sediments: The alkalinity of total CO_2 balance and methane production in the sediments of Lake Washington. *Limnology and Oceanography* 29:1218–1230.

Larsson, P. 1984. Transport of PCBs from aquatic to terrestial environments by chironomids. *Environmental Pollution Series* A34:283–289.

LaVesque, M. P., S. P. Mathur, and P. J. H. Richard. 1982. A study of physical and chemical changes in a cultivated organic soil based on palynological synchronization of subsurface layers. *Naturaliste Canada* 109:181–187.

LaZerte, B. D., and P. J. Dillon. 1984. The relative importance of anthropogenic versus natural sources of acidity in lakes and streams of Central Ontario. *Canadian Journal of Fisheries and Aquatic Sciences* 41:1664–1677.

Lazrus, A. L., E. Larange, and J. P. Lodge, Jr. 1970. Lead and other metal ions in United States precipitation. *Environmental Science and Technology* 4:55–58.

Lee, C. R., R. E. Hoepel, P. G. Hunt, and C. A. Carlson. 1976. Feasibility of the functional use of vegetation to filter, dewater, and remove contaminants from dredge material. Technical Report D-76-4. Environmental Effects Laboratory, U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, Mississippi.

Leisinger, T. 1983. Microorganisms and xenobiotic compounds. *Experientia* 39:1183–1191.

Lyman, W. J., W. F. Reehl, and D. H. Rosenblatt. 1982. Handbook of chemical property estimation methods. Environmental behavior of organic compounds. McGraw-Hill, New York, 960 pp.

Mason, C. F., and R. J. Bryant. 1975. Production, nutrient content, and decomposition of *Phragmites communis* and *Typha angustifolia*. *Journal of Ecology* 63:71–95.

Mathur, S. P., M. P. LaVesque, and P. J. H. Richard. 1981. The establishment of synchrony between subsurface layers and estimation of cultivated organic soils by a teleological method. *Canadian Journal of Soil Science* 62:427–431.

McAvoy, D. C., and J. W. Male. 1987. Modeling the fate of aluminum in a watershed system under acidified conditions. Report to the Massachusetts Division of Water Pollution Control, Westborough, Massachusetts, 199 pp.

McKnight, D., E. M. Thurman, R. L. Wershaw, and H. F. Hemond. 1985. Biogeochemistry of aquatic humic substances in Thoreau's Bog, Concord, Massachusetts. *Ecology* 66:1339–1352.

Mitsch, W. J., and J. G. Gosselink. 1986. Wetlands. Van Nostrand Reinhold, New York, 539 pp.

Neff, J. M. 1979. Polycyclic aromatic hydrocarbons in the aquatic environment: Sources, fates, and biological effects. Applied Sciences Publications, London.

Nichols, D. S. 1983. Capacity of natural wetlands to remove nutrients from wastewater. *Journal of Water Pollution Control F.* 55:495–505.

Nixon, S. W., and V. Lee. 1985. Wetlands and water quality. U.S. Army Technical Report Y-85, Washington, D.C.

Ogata, M., K. Fujisawa, Y. Ogino, and E. Mano. 1984. Partition coefficients as a measure of bioconcentration potential of crude oil compounds in fish and shellfish. *Bulletin of Environmental Contamination and Toxicology* 33:561–567.

Oliver, B. G., E. M. Thurman, and R. L. Malcolm. 1983. The contribution of humic substances to the acidity of colored natural waters. *Geochimica et Cosmochimica Acta* 47:2031–2035.

Olsen, E. T. 1983. The distribution and mobility of cesium-137 in *Sphagnum* bogs of northeastern North America. M.S. Thesis. University of Maine, Orono, Maine, 185 pp.

Oshwald, W. R. 1972. Sediment-water interactions. *Journal of Environmental Quality* 1:360–366.

Pakarinen, P., and K. Tolonen. 1976. Regional survey of heavy metals in peat mosses. *Ambio* 5:38–40.

Parr, J. F., and S. Smith. 1976. Degradation of toxaphene in selected anaerobic soil environments. *Soil Science* 121:57.

Patrick, W. N., and M. E. Tusneem. 1972. Nitrogen loss from flooded soil. *Ecology* 53:735–737.

Paulet, M., G. Konke, and L. Lund. 1972. An interpretation of reservoir sedimentation. I. Effect of watershed characteristics. *Journal of Environmental Quality* 1:146–150.

Peverly, J. H. 1985. Element accumulation and release by macrophytes in a wetland stream. *Journal of Environmental Quality* 14:137–143.

Phillips, R. E., K. R. Reddy, and W. H. Patrick. 1978. The role of nitrate diffusion in determining the order and rate of dinitrification in flooded soils. II. Theoretical analysis and interpretation. *Soil Science Society of America Journal* 42:272–280.

Pionke, H. B., and G. Chesters. 1973. Pesticide–sediments–water interactions. *Journal of Environmental Quality* 2:29–45.

Polunin, N. V. C. 1984. The decomposition of emergent macrophytes in freshwater. *Advances in Ecological Research* 14:115–166.

Rappaport, R. A., N. R. Urban, P. S. Capel, J. E. Baker, B. B. Looney, S. J. Eisenreich, and E. Gorham. 1985. "New" DDT inputs to North America: Atmospheric deposition. *Chemosphere* 14:1167–1173.

Reddy, K. R., W. H. Patrick, and R. E. Phillips. 1978. The role of nitrate diffusion in determining the order and rate of denitrification in flooded soils. I. Experimental results. *Soil Science Society of America Journal* 42:268–272.

Reicholf, J. 1976. The possible use of aquatic bird communities as indicators for the ecological conditions of wetlands. *Landschoft Stadt* 8:125–129.

Richardson, C. J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. *Science* 228:1424–1427.

Richardson, C. J., and P. E. Marshall. 1986. Processes controlling movement, storage, and export of phosphorus in a fen peatland. *Ecological Monographs* 56:279–302.

Rumer, R. R., Jr. 1977. Physical processes for pollutant transport in the water environment. Pages 297–322 *in* I. H. Suffet (ed.), Fate of pollutants in the air and water environment, part I. John Wiley and Sons, New York.

Ryden, J. C. 1978. The mechanism of phosphate adsorption on soil and anion exchange resin surfaces. *Journal of Soil Science* 28:72–81.

Schell, W. R. 1986. Deposited atmospheric chemicals. *Environmental Science and Technology* 20:847–853.

Schindler, D. W., R. Wagemann, R. B. Cooke, T. Ruszczynski, and J. Prokopowich. 1980. Experimental acidification of Lake 223, experimental lakes area: Background data and the first three years of acidification. *Canadian Journal of Fisheries and Aquatic Sciences* 37:342–354.

Schneider, I. F., and A. E. Erickson. 1972. Soil limitations for disposal of municipal wastewaters. Agricultural Experiment Station Research Report 195. Michigan State University, East Lansing, Michigan.

Schwarzenbach, R. P., and J. Westall. 1981. Transport of non-polar organic compounds from surface water to groundwater. Laboratory sorption studies. *Environmental Science and Technology* 15:1360–1367.

Seidel, K. 1970. Macrophytes and water purification. Pages 109–121 *in* J. Tourbier and R. W. Pierson, Jr. (eds.), Biological control of water pollution. University of Pennsylvania Press, Philadelphia.

Simmers, J. W., B. L. Folsom, Jr., C. R. Lee, and D. J. Bates. 1981. Field survey of heavy metal uptake by naturally occurring saltwater and freshwater marsh plants. U.S. Army Corps of Engineers Technical Report EL-81-5. Waterways Experiment Station, Vicksburg, Mississippi.

Singer, P. C. 1977. Influence of dissolved organics on the distribution, transport, and fate of heavy metals in aquatic systems. Pages 155–182 *in* I. H. Suffet (ed.), Fate of pollutants in the air and water environment, part I. John Wiley and Sons, New York.

Slater, H. J., and D. Lovatt. 1984. Biodegradation and the significance of microbial communities. Pages 439–485 *in* D. T. Gibson (ed.), Microbial degradation of organic compounds. Marcel Dekker, New York.

Sleat, R., and J. P. Robinson. 1983. The bacteriology of anaerobic degradation of aromatic hydrocarbons. *Journal of Applied Bacteriology* 57:381–394.

Steele, B. B., R. L. Bayne, Jr., and C. V. Grant. 1984. Environmental monitoring using populations of birds and small mammals: Analyses of sampling effort. *Biological Conservation* 30:157–172.

Suflita, J. M., J. A. Robinson, and J. M. Tiedge. 1983. Kinetics of microbial dehalogenation of haloaromatic substances in methanogenic environments. *Applied Environmental Microbiology* 45:1466–1473.

Thurman, M. 1985. Organic geochemistry of natural waters. Nishoff/Junk Publishers, Dordnecht, The Netherlands, 497 pp.

Tilton, D. L., and R. H. Kaldec. 1979. The utilization of a freshwater wetland for nutrient removal from secondarily treated wastewater effluent. *Journal of Environmental Quality* 8:328–334.

Urban, N. R., and S. E. Bayley. 1986. The acid–base balance of peatlands: A short-term perspective. Proceedings of the Muskoka Conference. *Water, Air, and Soil Pollution* 30(3–4):791–800.

Vestergaard, P. 1979. A study of indication of trace metal pollution of marine areas by analysis of salt marsh soils. *Marine Environmental Research* 2:19–31.

Weller, M. W. 1981. Freshwater marshes: Ecology and wildlife management. University of Minnesota Press, Minneapolis, Minnesota, 146 pp.

Whigham, D. F., and R. L. Simpson. 1976. The potential use of freshwater tidal marshes in the management of water quality in the Delaware River. Pages 173–186 *in* J. Toubier and R. W. Pierson, Jr. (eds.), Biological control of water pollution. University of Pennsylvania Press, Philadelphia, Pennsylvania.

Wieder, R. K., and G. E. Lang. 1986. Fe, Al, Mn, and S chemistry of *Sphagnum* peat in four peatlands with different metal and sulfur input. *Water, Air, and Soil Pollution* 29:309–320.

Willrich, T. L., and G. E. Smith. 1970. Agricultural practices and water quality. Iowa State University Press, Ames, Iowa, 415 pp.

Zafiriou, O. C., J. Joussot-Dubien, R. G. Zepp, and R. G. Zika. 1984. Photochemistry of natural waters. *Environmental Science and Technology* 18:358–371A.