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Indicators for Monitoring Biological Integrity of Inland, Freshwater Wetlands

A Survey of North American Technical Literature (1990-2000)

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by

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Notice

Part of the information in this report was compiled while one of the authors (T. Danielson) was at Duke University's Nicholas School of the Environment in Durham, NC. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

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Section 1. Introduction

This document has been written for wetland managers, researchers, and monitoring specialists. For wetland managers, it serves as a resource for identifying and understanding biological impacts that could result from regulated and unregulated activities in wetlands. For researchers and monitoring specialists, it facilitates interpretation of collected data by providing a context of what we already know.

In preparing this, our sole focus has been to update a literature review on the same topic sponsored and published previously by the USEPA (Adamus and Brandt 1990), see: http://www.epa.gov/owow/wetlands/wqual/introweb.html
As such, this document is *not* intended as stand-alone guidance for persons wishing to learn how to develop wetland biomonitoring programs, or for persons seeking to prioritize research. For additional wetland monitoring information, databases, publications, and guidance, readers should see:
http://www.epa.gov/owow/wetlands/bawwg/

This document grows from the recognition that in-depth knowledge of the most current research findings is imperative for developing and using scientifically-sound biological indicators of wetland condition. This document has the following important attributes:

- Literature from the period 1990-2000 is the primary focus.
- Literature from North American wetlands is the primary focus.
- Literature from inland freshwater wetlands is the primary focus.
- Literature on impacts to assemblages of species, rather than single species, is the primary focus
- Literature in peer-reviewed journals is referenced almost exclusively.
- Not every paper fitting the above descriptions could be reviewed. However, we believe this document -- based on review of over 1500 publications -- covers a majority of the relevant literature. The largest numbers of publications are cited in the Invertebrates and Vascular Plants sections of this document, but for these two topics we also excluded the largest numbers of relevant papers, due to limited time for review relative to the enormous number that were published in the last decade.

The past decade has seen predictable diversification of wetland research into an enormous array of subdisciplines and subtopics. A multitude of subjects previously unexplored and some never imagined have emerged in the literature. Our approach in preparing this document has been to emphasize wide coverage of the wetlands biological literature, rather than cover any particular topic or subtopic in depth. Because of the enormous number of studies that have been published, time constraints, and our stated goals for the effort, we have sought primarily to organize the recent literature in a helpful way, not to interpret or synthesize it.

The document is structured around 11 categories of human-related disturbances to which wetlands are commonly exposed (Table 1.1), and the effects of these “stressors” on the following groups, each the focus of a separate section: microbes, algae, vascular plants, invertebrates, fish, amphibians, and birds. This document assumes the reader is generally familiar with wetland terminology and ecological principles, as well as with terms and concepts that are associated with indices of biological integrity. For this reason no glossary is provided. This document does not attempt to summarize our understanding of each combination of stressor and biological group, but instead simply describes literature published since 1989 on each pairing of stressor with biological group. Information from saltwater wetlands, non-wetland systems, and foreign literature is included sparingly, mainly when it serves to highlight significant gaps in our understanding of the types of wetlands covered by this document.

Table 1.1. Human-related stressors addressed in this document

Enrichment, Eutrophication, Organic Loading, Reduced Dissolved Oxygen (DO). Interrelated increases in concentration or availability of nitrogen or phosphorus. Typically associated with excessive fertilizer application, livestock waste management, wastewater treatment systems, fossil fuel combustion, unmanaged urban runoff, and other sources. Includes increases in carbon, to the point where an increased biological oxygen demand reduces dissolved oxygen in the sediments and the water column and increases toxic gases (*e.g.*, hydrogen sulfide, ammonia).

Contamination Toxicity. Increases in concentration, availability, and/or toxicity of metals and synthetic organic substances. Typically associated with agriculture (pesticide applications), aquatic weed control, mining, urban runoff, landfills, hazardous waste sites, fossil fuel combustion, wastewater treatment systems, and other sources.

Acidification. Increases in acidity (decreases in pH). Typically associated with mining and fossil fuel combustion.

Salinization. Increases in dissolved salts, particularly chloride, and related to parameters such as conductivity and alkalinity. Typically associated with road salt used for winter ice control, irrigation return waters, stormwater, seawater intrusion (*e.g.*, due to land loss or aquifer exploitation), and domestic/industrial uses.

Sedimentation/Burial. Increases in deposited sediments, resulting in partial or complete burial of organisms and alteration of substrate. Typically associated with agriculture, disturbance of stream flow regimes, urban runoff, wastewater treatment plants, deposition of dredged or other fill material, and erosion from mining and construction sites.

Turbidity/Shade. Reductions in solar penetration of waters as a result of blockage by suspended sediments and/or overstory vegetation or other physical obstructions. Typically associated with agriculture, disturbance of stream flow regimes, urban runoff, poorly functioning wastewater treatment plants, and erosion from mining and construction sites, as well as from natural succession, placement of bridges and other structures, and resuspension by fish (*e.g.*, common carp) and wind.

Vegetation Removal. Typically associated with aquatic weed control, agricultural and silvicultural activities, channelization, bank stabilization, urban development, defoliation from airborne contaminants and other stressors included in this report, grazing/herbivory (*e.g.*, from muskrat, grass carp, geese, crayfish, insects), disease, and fire.

Thermal Alteration. Long-term changes (especially increases) in temperature of water or sediment. Typically associated with power plants, other industrial facilities, removal of shading vegetation, lowering of summertime water levels, and global climate warming.

Dehydration. Reductions in wetland water levels and/or increased frequency, duration, or extent of desiccation of wetland sediments. Typically associated with ditching, channelization of nearby streams, invasion of wetlands by highly transpirative plant species, outlet widening, subsurface drainage, global climate change, and ground or surface water withdrawals for agricultural, industrial, or residential use.

Inundation. Increases in wetland water levels and/or increase in the frequency, duration, or extent of saturation of wetland sediments. Typically associated with impoundment (*e.g.*, for cranberry or rice cultivation, flood control, water supply, waterfowl management) or changes in watershed land use that result in more runoff being provided to wetlands.

Other Human Influences. Increases in distance between, and reduction in sizes of, patches of suitable habitat (i.e., fragmentation). Increases in noise, predation from pets, disturbance from visitation, invasion by non-native species capable of outcompeting species that normally characterize intact communities; electromagnetic, ultraviolet (UV-B), and other radiation; and other factors not addressed above.

In addition to addressing the above for each biological group, this report briefly summarizes published information most relevant to monitoring the particular group. Within each group and under a subheading "Wetland Monitoring," recent information is compiled on spatial and temporal variation, techniques and equipment for monitoring the group, and biological metrics that have or have not been found to correlate with individual or cumulative disturbances to wetlands.

Ideally, it would be best to separate the published results by wetland type (e.g., Cowardin class, hydrogeomorphic class). Due to the lack of information on many groups, however, it was not practical to do so in this document. Thus, readers should be cautious when attempting to extrapolate the reported results.

This document was developed in four stages. First, Thomas Danielson identified, obtained, and prepared written reviews of relevant literature covering the period 1990-1996, with the exception of literature on wetland microbes and fish. Second, Alex Gonyaw (a student supervised by Paul Adamus in the Fisheries and Wildlife Department at Oregon State University) identified, obtained, and prepared written reviews of relevant literature covering wetland microbes and fish, plus updated the sections on the other groups, through literature published in 2000. Third, Paul Adamus edited the manuscript extensively and reviewed hundreds of publications that the co-authors had either not known about, or had not had time to review. Fourth, after re-readings and comments by the co-authors as well as external peer reviewers, Paul Adamus prepared the final document.

At every stage, potentially relevant literature for the years 1990-1999 was identified by (a) conducting keyword searches of computerized bibliographic databases, especially CAB Abstracts and Aquatic Sciences and Fisheries Abstracts, (b) reading through the tables of contents of a few especially relevant journals, (c) searching the internet for pertinent bibliographies, and (d) to a lesser extent, reviewing articles listed in these bibliographies and in the literature cited sections of relevant journal articles and books. Information and references from the parallel review of wetland biological studies specifically from Florida, prepared for the US EPA by Steve Doherty and others, were selectively incorporated.

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Section 2. Microbial Assemblages and Processes

2.1 Use as Indicators

This section addresses microbes that are closely associated with naturally-occurring wetlands. Included in this discussion are bacteria, protozoans, viruses, yeasts, and microscopic fungi (including mycorrhizae and hyphomycetes). Microscopic algae are discussed in Section 3. Like the other sections in this document, this section focuses almost entirely on research published since 1989. For a general discussion of the topic based on pre-1990 scientific studies, and for broader discussion including advantages and disadvantages of using microbial assemblages and processes as indicators of wetland integrity, readers should refer to Adamus and Brandt (1990).

Microbial organisms are omnipresent in wetlands, even living within the individual submerged roots of some wetland shrubs (Fisher et al. 1991). Through interactions with wetland plants and hydrology, wetland microbial assemblages can remove inorganic nutrients, heavy metals, dissolved organic carbon, particulate organic matter, and suspended solids from the water column and sediments (Mickle 1993), as well as play a key role in supporting food webs (Schallenberg & Kalff 1993) and influencing global climate change through their role in methanogenesis (Bartlett & Harriss 1993, Kumaraswamy et al. 2000). The presence of bacteria in "biofilms" on the enormous plant and detrital surface area in wetlands is fundamental to wetland ability to degrade complex organic contaminants (Hamilton *et al.* 1993, Taylor et al. 1996). Iron-oxidizing bacteria in roots of wetland plants also influence plant nutrition (Emerson et al. 1999). Production from indigenous bacteria may surpass production from algae in forested wetlands (Hudson et al. 1992). Many naturally-occurring bacteria inhibit waterborne pathogens. For example, bacteria that normally inhibit *Clostridium botulinum* type C (an organism that causes extensive waterfowl deaths) were found to be quite widespread in northern California wetlands (Sandler *et al.* 1998). A reduction in these inhibiting bacteria might lead to an increase in botulism poisoning of waterfowl.

Among the microbes, root fungi (mycorrhizae) are particularly important. Along with aquatic invertebrates, they are responsible for decomposing dead plant material, thus constantly renewing system nutrients (Tuchman 1993). Along with nitrogen-fixing bacteria, fungi supply nitrogen and/or phosphorus to wetland plant roots (Allen 1991, Wigand & Stevenson 1997, Pieher et al. 1998, Turner & Friese 1998, Lovell et al. 2000, Piceno & Lovell 2000). Indeed, lack of mycorrhizae in some constructed wetlands has been suggested as a reason for failure of plantings. Mycorrhizae occur widely in wetlands: an examination of 290 plants from 89 species of Connecticut wetland plants found that all species of mature plants as well as selected young plants on developing shorelines were colonized by mycorrhizal fungi (Cooke & Lefor 1998). This, plus surveys in Midwest wetlands (Turner & Friese 1998) and southeastern bottomland hardwoods (Jurgensen et al. 1997) and elsewhere suggest that mycorrhizae are common even among plants associated with fluctuating water, nutrient, and oxygen conditions.

Fungi also can influence the structure of vascular plant communities in wetlands. For example, lack of ectomycorrhizal fungi in beaver meadows (probably as a result of exclusion of fungi-spreading rodents by prolonged flooding) may prohibit the succession of these meadows into forested wetlands (Terwilliger & Pastor 1999). Reductions in ectomycorrhizal infection of willows in Alaska wetlands, as caused by herbivore browsing, caused a shift in plant community composition (Rossow et al. 1997). Southern boreal bogs and fens contain mostly mycorrhizal fungi that enable characteristic plant species to proliferate in these nutrient-poor ecosystems by accessing otherwise unavailable nutrient pools. In contrast, marsh vegetation mainly contains non-mycorrhizal fungi, possibly due to higher surface-water nutrient concentrations and fluctuating water levels (Thormann et al. 1999). In general, little is known about the effects of contaminant additions or other wetland alterations on mycorrhizae (Cairney & Meharg 1999).

Because microbial assemblages have so many important ecosystem functions, and because these functions are typically easier to measure than the taxonomic structure of the responsible microbial community, most literature on impacts describes effects on these functions rather than on the microbes themselves. To date, no North American studies have used microbial taxonomic or functional diversity or composition to assess the ecological conditions of a large series of wetlands, but use of microbes as indicators of aquatic system integrity has been considered generally (Lynn & Gilron 1992) and with reference to contaminants in particular (Maier et al. 2000).

2.2 Effects of Enrichment, Eutrophication, Reduced Dissolved Oxygen

Bacterial assemblages, with generation times as little as 15 minutes are well suited for detecting short-term nutrient pulses (Miyamoto and Seki 1992). In an Ohio marsh, experimental dosing with phosphate stimulated an increase in bacterial density (Willis and Heath 1993). Excessive enrichment can quickly allow anaerobic taxa to gain dominance. Microbial assemblages receiving agricultural nutrient inputs in part of the Florida Everglades were dominated by methanogens, sulfate reducers, and acetate producers (Drake *et al.* 1996). These bacteria flourish where porewater total phosphorus concentrations and conductivities are high (Drake *et al.* 1996). Excessive nutrients from agricultural operations may reduce the normal ability of wetland microbial assemblages to detoxify particular pesticides (Kazumi & Capone 1995, Chung et al. 1996, Entry & Emmingham 1996, Entry 1999). Although nitrogen additions to a riparian system briefly stimulated bacterial and fungal activity, long-term effects were perceived as negative, thus potentially compromising the ability of the system to remove nitrogen via denitrification (Ettema et al. 1999). Two constructed wetlands in Massachusetts had microbial biomass and potential denitrification that were within the range of variability of natural wetlands, owing at least partly to use of organic sediments in the construction of the wetlands (Duncan & Groffman 1994). Sediments in the Fox River/Green Bay watershed of Wisconsin were not toxic to *Photobacterium phosphoreum* despite having elevated ammonia (1.3-54.4 mg/L) (Ankley *et al.*1990). Some bacteria (termed nitrifiers) play a key role in wetland food webs by converting ammonium and nitrite to nitrate, which supports algae and vascular plants (Groffman et al. 1992). This nitrifying capacity is influenced by the plant species with which the bacteria are associated (the plants may secrete antibacterial substances) and not with the cation exchange capacity of the plant litter (Eriksson & Andersson 1999).

In New York, experimental additions of particulate detritus derived from the most common submerged macrophyte (*Vallisneria americana*) and wetland plant (*Typha angustifolia*) to Hudson River water did not result in increases in bacterial productivity. In contrast, additions of dissolved organic carbon derived from these same plants consistently yielded large increases in bacterial production (Findlay *et al.* 1992). Growth response of bacterial colonies in streams may depend on timing and source of natural leachates from local plants as well as on sources of dissolved organic carbon from further upstream. Growth of bacterial assemblages in streams exhibited 'generalist' characteristics in headwater reaches and 'specialist' characteristics farther downstream (Koetsier et al. 1997).

Decomposition rates, which generally reflect microbial activity, increased along a eutrophication gradient from a bog, a poor fen, a wooded moderate-rich fen, a lacustrine sedge fen, a riverine sedge fen, a riverine marsh, and a lacustrine marsh in southern Alberta (Thormann *et al.* 1999). Although ammonium fertilizers have been thought to potentially increase methane emissions from wetlands (due to effects on particular bacterial assemblages), evidence to the contrary was reported by Bodeller et al. (2000).

Colonization of wetland plants by mycorrhizal fungi is sometimes greater in less fertile (low phosphorus) wetlands (Wetzel & Van der Valk 1996, White & Charvat 1999), but nutrient levels are probably not a major influence. Also, mycorrhizae can be restricted by low oxygen conditions typical of some eutrophic wetlands (Johnson et al. 1992, Johnson 1993, Cantelmo & Ehrenfeld 1998).

2.3 Effects of Contaminant Toxicity

Bacteria have a variety of responses to contaminants, ranging from direct utilization of toxins as energy sources, to being being harmed or unaffected. Crude oil, leaded gasoline, and the herbicide 2,4,5-T reduced taxonomic and/or functional diversity of microbial communities (Atlas et al. 1991). High concentrations of iron, manganese, magnesium, calcium and nickel in water from an abandoned coal mine reduced the diversity of fungi that decompose leaf litter. This, plus a shift in fungal species composition, reduced the leaf decomposition rates (Bermingham et al. 1996). Several other contaminants, by inhibiting microbial and invertebrate decomposers, can slow the rate of decomposition of wetland vegetation at the end of each growing season (Schultheis and Hendricks 1999), potentially causing anoxia in wetlands as springtime temperatures rise. However, the species composition of fungal decomposers in streams exposed to the insecticide methoxychlor did not differ significantly from that in unexposed streams (Suberkropp & Wallace 1992). A single application of the pesticide diquat reduced protozoan species richness, net productivity, community respiration, and enzyme activities. Both photosynthetic and non-photosynthetic taxa were affected by diquat (Melendez et al. 1993).

Antibacterial agents (e.g., tetracyclin) in domestic wastewater, such as from leaky septic systems and feedlot runoff, have the potential for inhibiting microbial assemblages and thus harming fundamental ecosystem processes. Some instances have been noted of evolution of resistant strains in nature, thus posing potential threats to human health (DePaola 1995).

Some microbial assemblages tolerate or even biodegrade particular contaminants. Such assemblages can be naturally selected and flourish in some wetlands that are only mildly or moderately contaminated. Contaminants that can be processed when at low-moderate concentrations by microbes include copper (Farago and Mehra 1993), mercury (Marvin-Dipasquale & Oremland 1998), selenium (Steinberg & Oremland 1990, Azaizah 1997), cadmium (Sharma et al. 2000), manganese (Sikora et al. 2000), and petroleum (Nyman 1999, Megharaj et al. 2000). Microbial assemblages attached to plant roots also help wetland plants take up some metals (e.g., selenium, mercury) contained in runoff (Souza et al. 1999). Sulfate-reducing bacteria play a major role in detoxifying some contaminants, and may be most diverse in oxygen-containing environments (Minz et al. 1999), but other functional groups of bacteria can also be important (Webb et al. 1998). A wetland fed by leachate from an abandoned landfill in Washington supported large populations of total and fecal coliforms, fecal streptococci, the opportunistic pathogens *Pseudomonas aeruginosa, Staphylococcus aureus, Enterococcus faecalis*, and the anaerobe *Clostridium perfringens* (Boening and Vasconcelos 1997). A Georgia tidal wetland containing elevated levels of mercury and PCB's showed little evidence of adverse impact to fungal communities; elevated nutrient concentrations associated with the contaminants may have somewhat offset any toxic effects occurring at the population level (Newell & Wall 1998).

Compensatory mechanisms within microbial assemblages sometimes cloud the response to complex mixtures of toxicants. In aquatic microcosms with continuous dosing, the herbicide atrazine was found to increase microbial richness and biomass, whereas copper reduced these attributes and chlorpyrifos and nutrient-rich chlorinated effluent had no detectable effect (Pratt et al. 1993).

2.4 Effects of Acidification

Microbes are highly sensitive to pH (James 1991). Taxa richness and species composition appear to be more sensitive to acidity than biomass and net oxygen metabolism. Bacterial abundance and richness of protozoan assemblages has been shown to decline below a pH of 5.3, and shifts in taxa composition can occur at higher pH's (Niederlehner & Cairns 1990). Slow rates of plant litter decomposition can indicate acidic conditions, as noted in a West Virginia wetland (Kittle *et al.* 1995). Wetlands with low acidity and high salinity are generally at greatest risk for outbreaks of *Clostridium botulinum,* the cause of avian botulism poisoning (Rocke and Samuel 1999). Acidification of soils can diminish populations of mycorrhizae that are crucial to plant nutrition (Hutchinson et al. 1999).

Occasionally, some microbial assemblages respond positively to moderately acidic conditions (Brenner 1995). Acidity enhances the availability of metals that are a source of energy for many bacteria. Wetlands that have been acidified, or which naturally have more acidic conditions, support a characteristic assemblage of iron-oxidizing bacteria (Emerson 1999). Dissolved carbon levels, which often parallel acidity levels, can influence microbial population densities even more than can acidity (Fisher *et al.* 1991).

2.5 Effects of Salinization

Diverse bacterial assemblages are key to cycling of energy in many naturally saline wetlands (Zahran 1997). Bacterial mineralization (i.e., increased bioavailability) of dissolved organic nitrogen declines as salinity increases along a freshwater to seawater gradient (Stepanauskas *et al.* 1999). Decomposition of cellulose gradually declines along a salinity gradient, although at high salinity (seawater strength) decomposition can increase greatly (Mendelssohn *et al.* 1999). Salinities of greater than about 300 g/L can inhibit the ability of microbial assemblages to detoxify toxic forms of selenium (Steinberg & Oremland 1990).

2.6 Effects of Temperature

Methanogenesis by microbes is much greater at warmer temperatures (Boon and Mitchell 1995, Sorrell *et al.* 1997), as is microbial degradation of some detergents, e.g., alkybenzenesulfonates (Inaba 1992). Microbial communities in wetlands containing hot springs exhibit a tremendous degree of genetic diversity (Hugenholtz et al. 1998).

2.7 Effects of Sedimentation, Burial

Fine sediments generally contain larger populations of microbes than do coarse sediments, because they provide additional surfaces for attachment and protection from predators (Davies & Bavor 2000). However, application of unenriched and phosphorus-enriched fine clay suppressed decomposition of *Sparganium eurycarpum* decay by about 6-8% over 117 days (Vargo *et al.* 1998). A single application of coarse sediments to cat-tail (*Typha latifolia)* litter inhibited decomposition by 10% over 470 days.

2.8 Effects of Vegetation Removal, Shade

Bacterial biomass production was unaffected by light intensity under the conditions used in one series of laboratory experiments (Neely and Wetzel 1997). Fungal biomass was found to be greater than bacterial biomass throughout the year in an Oregon riparian system (Griffiths et al. 1997). Colonization of wet prairie herbs by fungal mycorrhizae in Oregon was not significantly correlated with prior type of intensity of land use (Ingham & Wilson 1999). Widespread removal of host plants by prolonged flooding reduced the extent of colonization of remaining plants by mycorrhizae (Ellis 1998).

Decomposition of various wetland macrophytes did not differ significantly among grazed and ungrazed sites in a German river delta (Ibanez *et al.* 1999), but mowed sites had less colonization by mycorrhizae (Titus & Leps 2000). Decomposition of cotton strips was significantly greater following logging of a Michigan forested wetland (Trettin and Jurgensen 1992). Removal of vegetation and the upper soil layer from a Florida Everglades site resulted in increased activity of microrrhizal fungi for at least 2 years after disturbance (Aziz et al. 1995). However, removal of streamside vegetation in Illinois resulted in fewer number of decomposer fungal species per unit substrate and reduced colonization of substrates (Metwalli & Shearer 1989). Denitrification rates also have been found to differ between grass vs. forested riparian areas (Groffman et al. 1991).

2.9 Effects of Dehydration, Inundation

Changes in water volume in wetlands have little effect on individual bacteria directly, but resulting changes in nutrient levels and salinity, brought on by dilution or concentration of elements, can cause profound changes. Bacterial densities in wetlands often increase shortly after runoff events that bring in nutrients (White *et al.* 1998). Following the end of a drought, bacterial densities in a clearwater lake exposed to acidic precipitation were unchanged, whereas in a darkwater (naturally stained) lake, bacterial densities declined possibly as the result of interactive effects of high dissolved organic carbon concentrations and reduced pH (James 1991). Leaf litter decomposed faster, and nitrogen was mineralized more readily, in a Louisiana crayfish pond where flooding was manipulated than in natural and impounded wetland forests (Conner and Day 1991). Decomposition rates were greater in drained than undrained pocosins (peat wetlands) in North Carolina (Bridgham et al. 1991). In Manitoba, decomposition rates were found to be higher in a flooded treatment than in the control treatment for some plant species (van der Valk *et al.* 1991), but not others (Wrubleski *et al.* 1997).

Flooding of wetlands stimulates microbial activity that potentially increases conversion of inorganic mercury to the much more toxic methyl mercury form (Kelly *et al.* 1997, Heyes *et al.* 1998). Either increases or decreases in wetland water levels, if occurring over sustained periods, can increase bacterial release of methane from organic sediments (Freeman et al. 1996, Brown & Clair 1998). However, lowering the water level below the sediment surface in part of the Florida Everglades caused the wetland to shift from a methane source to a methane sink (Happell et al. 1993).

Although effects of changing water levels on denitrification have not been studied in prairie wetlands, two recent landscape-scale studies of Saskatchewan fields (Elliott and de Jong 1992, van Kessel et al. 1993) highlight the key role of soil moisture. Soil water content was found to be the most dominant factor controlling denitrification activity, followed by the concentration of ammonium, total soil respiration, and nitrate (van Kessel et al. 1993). Microbial assemblages that support denitrification develop rapidly in newly created wetlands (Duncan and Groffman 1994).

Some studies suggest that colonization of plant roots by vesicular arbuscular mycorrhizal (VAM) fungi can be influenced by wetness (Cooke et al. 1993, Cantlemos and Ehrenfeld 1999), whereas others suggest it is only mildly influenced by wetness (Aziz et al. 1995, Turner et al. 2000). The *species composition* of VAM fungi was found to vary only slightly by water depth when plant species was held constant (Miller & Bever 1999). No VAM species was confined to the wettest parts of two wetlands that were studied in Florida (Miller & Bever 1999). VAM *colonization* has found to be less where surface water depth and persistence are great (Rickerl et al. 1994,Wigand et al. 1998, Miller & Sharitz 2000, Miller 2000).

2.10 Wetland Monitoring

Spatial and Temporal Variation

Bacterial abundance and productivity can vary more than an order of magnitude over an annual cycle. Bacterial abundance during warmer periods can be measurably affected by protistan herbivory, although much of this herbivory is respired (Johnson and Ward 1997). In an Alabama wetland, bacterial productivity per mg dissolved organic carbon (DOC) in spring decreased compared to winter, while dissolved organic carbon concentrations increased over this period (Mann 1998). In an Ohio fen, microbial growth rates and cell density peaked in late July and then decreased until at least December (Gsell *et al.* 1997). Surface DOC within a southern wetland varied seasonally, with greatest fluctuations in concentrations through the summer and autumn during intensive macrophyte growth and bacterial production (Mann and Wetzel 1995). Methane emission in a Maine peat bog reached its lowest point in winter (Roslev and King 1996). Methane production in organic-rich wetlands is related more to organic-chemical components of the peat than to the activity of coincident sulfate-reducing bacteria (Yavitt and Lang 1990). In salt marshes, the species composition of diazotrophic bacteria assemblages (which provide nitrogen to plant roots) despite large acute variations in available carbon (Piceno & Lovell 2000b). Bacterial and fungal populations can occur in deep alluvial sediments over 1 mile from a river channel, but are greater closer to the channel (Ellis et al. 1998). Spatial variation in nitrification rates in a wetland was mostly associated with differences in emergent plant species composition (Eriksson & Andersson 1999). One study found substrate type to have a greater influence than local geography on microbial taxonomic composition (Goodfriend 1998).

Techniques and Equipment

Because of the highly dynamic nature of microbial assemblages, obtaining samples that are spatially and temporally representative of the community's taxonomic composition and density can be a daunting or impossible task (Kinkel et al. 1992). Instead, diversity of *functional processes* is often measured with much less effort. Or, microbial taxa can be grouped by presumed physiological tolerances, nutritional versatility, genetic distinctiveness, or other factors, prior to analyzing the data (e.g., Atlas 1991). A technique using rRNA-targeted oligonucleotide probes was found to be reliable for characterizing functional composition of microbes in salt marsh sediments (Edgecomb et al. 1999, Frischer et al. 2000). A method for assessing populations shifts within functional groups of microbes (such as change in proportion of nitrogen-fixing microbes) in response to pollution also is described by Chelius & Lepo (1999). Isotopes are increasingly being used to identify key sources of nutrition for microbial assemblages in wetlands, thus better defining the temporally and spatially variable roles of wetland algae vs. macrophytes (Creach et al. 1999). Microbial taxa responsible for sulphate reduction and methane oxidation can be identified by use of stable carbon (^{13}C) isotopes (Boschker et al. 1998).

To characterize wetland microbial diversity according to physiologic characteristics, API and BIOLOG tests can be performed on samples (Bagwell et al. 1998). Also, DNA, protein, and lipid synthesis can be assessed by measuring thymidine, leucine, and glucose incorporation (Gsell et al. 1997). A review of commonly used microbial toxicity tests (Van Beelen & Doelman 1997) concluded that (1) respiration rate per unit of biomass is a more sensitive indicator of toxic effects than the respiration rate or the amount of biomass alone, (2) autotrophic nitrification and acetylene reduction tests can be sensitive to toxics when short incubation times are used, (3) the nitrogen mineralization, denitrification and many enzymatic tests are often not very sensitive to effects of toxics, and (4) urease activity is a relatively sensitive enzymatic test for contaminants. Much information on microbiological techniques is summarized by Maier et al. (2000).

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Section 3: Algae

3.1 Use as Indicators

From a habitat perspective, algae are commonly grouped as phytoplankton (algae suspended in the water column), metaphyton (unattached and floating or loosely associated with substrata), benthic algae (attached to substratum), and epiphytic algae (attached to plants). Like the other sections in this document, this section focuses almost entirely on research published since 1989. For a general discussion of the topic based on pre-1990 scientific studies, and for broader discussion including advantages and disadvantages of using algal assemblages and processes as indicators of wetland integrity, readers should refer to Adamus and Brandt (1990) and to EPA's web pages on use of algae in wetland monitoring: http://www.epa.gov/owow/wetlands/bawwg.

As a source of energy for invertebrates and higher trophic levels, algae often are more important than vascular plants, at least during the summer when temperature and light conditions are most favorable (Hanson & Butler 1990, Hargeby et al. 1991, Neill & Cornwell 1992, Murkin et al. 1992, Peterson & Deegan 1993, Vymazal 1994, Campeau et al. 1994).

In the past decade, studies that have used assemblages of algae specifically to indicate condition of a large series of wetlands have been conducted, for example, in Montana (Apfelbeck 1998), the Midwest (Mayer & Galatowitsch 1999), and southwestern Maine (*in preparation*). Some of these studies are detailed at http://www.epa.gov/owow/wetlands/bawwg/case.html .

Of particular note is the book *Algae and Element Cycling in Wetlands* (Vymazal 1994), and the book *Algal Ecology* (Stevenson et al. 1996), especially the chapter on wetland algae by Goldsborough and Robinson (1996). Also, literature specifically on algae in prairie wetlands was reviewed by Crumpton (1989) and Adamus (1996). Information on use of algae in the development of biocriteria, and/or on the tolerances of many algal taxa to eutrophication, acidification, and/or herbicides, is discussed or compiled by Shubert 1984, Dixit *et al.* 1991, Whitton et al. 1992, McCormick and Cairns 1993, Dixit and Smol 1994, Harper 1992, van Dam *et al.* 1994, Patrick and Pavalage 1994, Rosen 1995, Stevenson et al. 1996, Pan & Stevenson 1996, McCormick & Stevenson 1998, and Stevenson et al. 1999.

3.2 Effects of Enrichment, Eutrophication, Reduced Dissolved Oxygen

Processes

Algal production is often constrained by one or two macronutrients (Hannson 1992), most notably phosphorus and nitrogen. For example, Everglades wetlands are naturally limited by phosphorus (Browder et al. 1994, Pan et al. 1999, 2000), whereas arctic systems respond to either nitrogen, phosphorus (Lock *et al.* 1989) or both (Bowden *et al.* 1992). A number of studies involving experimental dosing of actual or simulated wetlands with nutrients have been conducted in the past decade, and algal responses have been monitored (Table 3.1).

Table 3.1. Examples of nutrient response studies involving wetland algae

STUDY SITE	DURATION	NUTRIENT CONCENTRATION	REFERENCE
Lab 2 m X 19 cm	61 days	0, 0.1, 0.5, 1, 2, 5, 10, 20, 50, 100 µg/L PO_3	Bothwell 1989
Tundra stream	6 years	10 µg/L P, 100 µg/L N	Bowden *et al.* 1992
Marsh	2 months	2, 3, 5, 6, 7, 27, 51, 76 112 µg/L Dissolved P	Grimshaw *et al.* 1993
Everglades Slough	5 months	0.4, 0.8, 1.6, 3.2, 6.4, 12.8 µg/L NaH_2PO_4	McCormick and O'Dell 1996

Whereas in some lakes and streams algal mats indicate eutrophication, in the Florida Everglades algal mats with *Utricularia* spp. are viewed as indicators of health (McCormick and Stevenson 1998, Craft et al. 1995, Rader and Richardson 1992). With increasing nutrient loading, however, the polysaccharides that hold algal mats together disintegrate (McCormick et al. 1997, McCormick and Stevenson 1998, Craft et al. 1995, Rader and Richardson 1992). Although the mats themselves dissipate, the species responsible for the polysaccharides typically remain, unless affected by other variables (Rader and Richardson 1992). In some cases where nutrient loading continues, desmid species that construct the mats are replaced by more nutrient tolerant species. Craft et al. (1995) found that as algae mats dissipated, *Chara* spp. became dominant.

Effects on Species Richness

Eutrophication can lead to the simplification of algal communities, especially those already in a mesotrophic environment. In streams, diatom communities respond to organic enrichment with decreased species richness, diversity, and evenness (Steinman and McIntire 1990). When phosphoric acid was added to an Alaskan river, for example, species diversity and evenness declined (Bowden *et al.* 1992).

Effects on Species and Functional Group Composition

Generally, phytoplankton respond quickly to small, repeated additions of nutrients (Jorgenson *et al.* 1992, O'Brien *et al.* 1992, Olsson *et al.* 1992, Gabor *et al.* 1994), as does epiphyton (Wetzel 1990) . However, phytoplankton may respond more strongly than epiphyton to a single dose of nutrients added in the spring (Gabor *et al.* 1994). Metaphyton respond even more slowly, but effects are more enduring. At least in deeper wetlands and lakes, the response of benthic algae to nutrients is relatively muted (Murkin *et al.* 1994), but nonetheless measurable (Lowe and Pan 1994).

Particular algal taxa seem to respond more quickly than others to nutrient enrichment. For example, *Chlorella*, *Spirogyra*, and *Anabaena* are often found in nutrient-rich waters, whereas *Cladophora* and *Ankistrodesmus* sometimes occur in less eutrophic situations (Hosseini and van der Valk 1989a). When phosphoric acid was added to an Alaskan river, the dominant taxon *Hannea arcus* declined, as did *Fragillaria*, whereas numerous taxa belonging to Achnanthese, Cymbella and Eunotia increased (Miller *et al.* 1992). Among 30 Canadian lakes, total phosphorous and water transparency (Secchi depth) were found to explain a large proportion (85%) of the species variance (Agbeti 1992).

Among wetlands of the United States, algal species response to nutrients has probably been best studied in the Florida Everglades. During a nutrient addition experiment there, periphyton declined strongly after one year of treatment, and their species composition shifted (McCormick *et al.* 1997). Chlorophytes (filamentous green algae) and diatoms replaced the normal assemblages of cyanobacteria (blue-greens), whereas vascular plants showed no response (Rader & Richardson 1992, Daoust and Childers 1997).

The decline in the Everglades specifically of periphyton was also documented by Jensen *et al.* (1995), and its cause has been attributed primarily to phosphorus rather than nitrogen (Vymazal *et al.* 1994). Along a 14 km nutrient gradient in the Florida Everglades, shifts in periphyton community composition were related strongly to decreasing distance from canal discharges and to increasing total phosphorus (TP) ranging from 10 (8 km from canal discharges) to 150 µg/L (near discharges) (McCormick *et al.* 1996). In particular, diatom species indicative of low TP (e.g., *Anomoeoneis vitrea, Mastogloia smithii*) were consistently replaced by eutrophic indicator species (e.g., *Gomhonema parvulum, Nitzschia amphibia*) at TP concentrations between 10 and 20 µg/L. Another study along the same gradient found declines in the relative abundance of five oligotrophic indicator species (*Scytonema hoffmanii, Shizothrix calcicola, Oscillatoria limnetica, Cymbella lunata* and *Mastogloia smithii*) at 10 µg TP/L (McCormick and O'Dell 1996). The abundance of *Anomoeoneis serians* displayed a less severe decline at this TP level. Cyanobacteria/diatom assemblages dominated stations with less than 10 µg TP/L, whereas intermediate TP loading levels (3.2-6.4 g P/m^2/yr) encouraged the establishment of floating *Spirogyra* sp. mats. As TP levels increased to approximatey 40-50 µg/L other taxa became dominant including *Nitzschia* sp., *Lyngbya aestuarii* and *Oscillatoria minima*. Independently, Raschke (1993) found similar shifts in taxonomic composition in the Everglades below 20 µg/L TP.

The exact nutrient that contributes to algal community shift is often difficult to identify due to correlations among many nutrients (McCormick and O'Dell 1996). Nonetheless, some studies (Harper 1992) have reported that diatoms seem to dominate at lower temperatures and when phosphorus (P) but not silica (Si) is limiting, whereas green algae may dominate at higher temperatures with moderate or low N:P and Si:P ratios; Cyanobacteria (blue-green algae) typically dominate at higher temperatures and at low N:P ratios, and often characterize highly enriched waters (Hughes and Paulsen 1990, Murkin et al. 1991, Murkin et al. 1994, Biggs 1995). In the Florida Everglades, phosphorus has the largest impact on algal assemblages, followed by nitrogen (N) and iron (Fe) (McCormick & O'Dell 1996, McCormick et al. 1998). In enriched areas of the Everglades, nitrogen, other nutrients, and/or light play a larger role in limiting growth (Vaithiyanathan and Richardson 1997, McCormick and Stevenson 1998). In nitrogen limited areas, increases in *Rhopalodia gibba* and blue-green algae (*Nostoc)* and typical (Vaithiyanathan and Richardson 1997).

Other chemical constituents of water, particularly pH-altering bicarbonates, can regulate the response of algae to nutrient additions (Fairchild and Sherman 1993). In a Canadian softwater oligotrophic lake, *Stigeoclonium* sp., *Scenedesmus* sp., *Cryptomonas* sp., *Euglena* sp. and *Rhodomonas* sp. increased in relative abundance when carbonate ions were naturally abundant during the addition of nitrogen (.15 mol $NaNO_3$) and phosphorus (.015 mol Na_2HPO_4), whereas *Mougeotia* sp., *Oedogonium* sp., *Nostoc* sp. and *Anacystis* sp. decreased (Fairchild *et al.* 1989, 1989a). Species responses varied depending on whether nitrogen, phosphorus or bicarbonate were supplied, indicating that limiting nutrients were species specific. Excessive concentrations of nitrogen, in the form of ammonia – can be directly lethal to algae. In Michigan, ammonia contaminated sediments (1.3-54.4 mg/L ammonia) from 13 sites were acutely toxic to *Selenastrum capricornum (*Ankley *et al.* 1990). In addition, physical factors play a role in the response of alge to nutrients. In Pool 8 of the Mississippi river -- a predominantly lacustrine system with various lentic and slow water lotic microhabitats -- algal communities appear to be shaped by water flow, rather than strictly regulated by nutrient levels (Lange and Rada 1993).

Shifts in nutrients can alter macroinvertebrate populations, leading to a change in the balance between those that consume algae vs. those that consume vascular plants. This shift in trophic levels can mask the effects of nutrient additions (Irvine *et al.* 1989). Enrichment also can shift a stable epiphyton-dominant system ("open wetland") to metaphyton dominance ("sheltered wetland") if macrophytes remain sufficiently abundant to provide a substrate for metaphytic algae (McDougal *et al.* (1997). Development of the phytoplankton-dominant "lake wetland" state presumably occurs only when other algal and macrophytic competitors for nutrients are few. Metaphytic algae may increase in dominance due to their limited palatability to micro and macroinvertebrates (Neill and Cornwell 1992).

The use of algal species composition to predict ambient TP concentrations is limited by extensive variation in time and space of the TP concentrations (Chambers *et al.* 1992, France and Peters 1992). Nutrients appear to vary even more than pH (Battarbee 1990, 1999).

Effects on Density or Biomass

In response to enrichment, algal biomass increases quickly (Humphrey and Stevenson 1992, Dodds et al. 1998) -- more quickly than does biomass of submerged vascular plants (Crumpton 1989, Klarer and Millie 1992). For example, in a lacustrine wetland in Manitoba, algal biomass and density were strongly correlated to the degree of wetland enrichment (Murkin *et al.* 1991b). In the Florida Everglades, phosphorus concentrations in periphyton corresponded closely with concentrations in the water across the range of 3 to 112 μg TP/L, implying substantial uptake of phosphorus by the algae (Grimshaw *et al.* 1993). Uptake by Everglades epiphyton, vascular plants, and sediments resulted in a ten-fold decline in ambient phosphorus levels (Grimshaw *et al.* 1993).

The increase in algae that results from an increase in nutrients depends largely on the intensity of algal grazing by invertebrates (Harris 1996, Allan 1995, Bourassa and Cattaneo 1998, Mazumder et al. 1989, Paul et al. 1989, Mulholland et al. 1991). For example, in open water systems, large cladocerans can suppress algal biomass to low levels despite excess nutrients. Additions of phosphorus to an arctic river resulted in a net decrease in epilithic algae due to an extreme increase in grazers (Miller *et al.* 1992). Similarly, when nutrients were added to a Canadian marsh, net algal production (predominantly epiphyton) declined due to grazing from invertebrates (Hann and Goldsborough 1997). However, in a series of ponds treated with phosphate, algal biomass was higher – despite the abundance of a snail -- than those where nitrate was added (McCormick and Stevenson 1989).

Shading also influences the magnitude and type of response of algae to nutrients. In 11 Canadian lakes, epiphyton biomass increased up to 39 μg total phosphorus/L, after which it declined due to shading from phytoplankton (Lalonde and Downing 1991). Metaphyton also create shade, and respond quickly to phosphorus additions (e.g., 179 mg-P/m^3)(Wu and Mitsch 1998).

3.3 Effects of Contaminant Toxicity

Processes

Much of the literature on algal sensitivity to chemical contaminants, as revealed by experimental dosing (phytotoxicity testing), is summarized by Lewis (1995). Many algal species are more sensitive than vascular plants to contaminants, especially contaminants that interfere with photosynthesis. Common pollutants often elicit responses from algae that are well below the levels that affect vascular plants and other groups (Niederlehner and Cairns 1990, Lewis et al. 1998). Algal sensitivity to contaminants can also be less in eutrophic environments (Chen 1989, Wangberg and Blanck 1990).

Prior exposure to a contaminant can alter the response of individual cells to a new contaminant (Niederlehner and Cairns 1990), and consequently entire populations can develop resistance to a chemical after chronic exposure (Blanck and Wangber 1988). Algal communities can recover from exposure to toxic pollutants by means of their physiological or genetic adaptations to current conditions, or by undergoing a slow successional change after the disturbance ends, which sometimes returns the community to its predisturbance condition (Steinman and McIntire 1990).

Effects on Biomass

If algal biomass is the only metric considered, then some studies suggest that the stimulatory effect of compounds remaining in treated wastewater mixtures sometimes outweighs the toxic effects. For example, Lewis et al. (1998) examined the response of *Selenastrum capricornutum* (a freshwater green algae) to *in vitro* additions of different effluent from two cities, one naval air station, three forest product plants, two agro-chemical industries, one synthetic fibers industry, and one steam power generation plant. Algal biomass was stimulated in all but one forest product effluent and one city effluent.

Effects on Species Composition, Growth

Herbicides often cause a shift from large, filamentous green algae (chlorophytes) to smaller diatoms and cyanobacteria species (Gurney and Robinson 1989). Ironically, algal blooms can occur in wetlands after herbicides are applied to kill vascular plants, because a reduction in shade from vascular plants can trigger increases in benthic algae and metaphyton (Adamus 1996).

After testing 20 herbicides, 2 insecticides, and one fungicide on many algal species, Peterson and others (1995) reported adverse effects from 9 of the pesticides (particularly 5 triazine herbicides). The fungicide propiconazole and the herbicides picloram, boromoxynil, and triclopyr were relatively harmless to algae. In another study, stream periphyton did not appear to be adversely impacted by a 12-hour exposure to hexazinone (Kreutzweiser *et al.* 1995). The insecticide, Fenitrothion™, caused significant decreases in growth among 12 phytoplankton species (Kent *et al.* 1995). Atrazine and bifenthrin had deleterious effects on algal populations (Hoagland et al. 1993).

Toxic levels of copper, lead, and zinc have been documented to cause a decline of many algal species and an increase of *Rhizosoenia eriensis* in a contaminated lake (Deniseger *et al.* 1990), as well as affect algal metabolic processes (Hill et al. 1997). Phytoplankton from an Ohio lake were more sensitive to copper toxicity during the summer and fall than in the spring, except for *Crythophyta* which was strongly sensitive to copper throughout the year (Winner and Owen 1991). Polynuclear species of aluminum may be very toxic to algae and may represent a significant proportion of the aqueous aluminum at some conditions of low pH (Hunter and Ross 1991). However, in a stream experiment the addition of aluminum increased densities of diatoms, green algae, and blue-green algae (Genter and Amyot 1994). Specifically**,** the diatom *Achnahes minutissima*, the green alga *Cosmarium malanosporum*, the filamentous blue-green alga *Schizohrix calcicola*, and *Navicula* sp. diatoms all experienced increased growth in response to elevated aluminum (Genter 1995). Another study found *Navicula* sp. to be unaffected by the addition of aluminum (Planas *et al.* 1989).

3.4 Effects of Acidification

Processes

Algae are affected by acidification as a result of (a) direct toxicity (Baker and Christensen 1990, Fairchild and Sherman 1993), (b) indirect toxicity, from some metals that are mobilized or made more available by changes in acidity (Genter and Amyot 1994, Schindler 1990, Kingston *et al.* 1992), and (c) changes in competition with, and predation from, organisms that are less sensitive (Elwood and Mulholland 1989, Schindler 1990, Locke and Sprules 1994, Feminella and Hawkins 1995). Changes resulting from algal sensitivity to acidification can be traced through entire food webs (Havens 1992).

Effects on Species Richness

Either extreme of acidity (too acid or too basic) can dimish species richness of algal communities. Among 36 lakes in the Upper Midwest, diatom community diversity and richness were lowest in the most alkaline, plankton-dominated lakes and in lakes dominated by *Melosira* sp. (Cook and Jager 1991).

Effects on Species Composition

Logically, the losses of algae in acidified waters are proportionally greater for acid-intolerant species (Havens and Heath 1990, Pillsbury and Kingston 1990). Species composition is perhaps a better indicator of acidification than is biomass (Dixit and Smol 1989).

Acidification, especially when accompanied by increased availability of aluminum, can result in an increase in green algae (e.g., *Mougeotia*, *Temnogametum)* and decrease in cyanobacteria (e.g., *Lyngbya* and *Anabaena)* and diatoms (e.g., *Achnathes)* (Turner *et al.* 1991, Dixit *et al.* 1991b). In a series of lake acidification experiments, green algal mats greatly increased as pH decreased from 6.0 to 5.1 (Baker and Christensen 1991). *Mougeotia* sp. also grew extensive mats in a stream experiment that maintained a pH of 5.0 for extended periods of time (Elwood and Mulholland 1989).

During acidification, the phytoplankton community often shifts from smaller to larger types with an accompanying increase in transparency of the water (Havens 1991b). Long, thin, pennate diatoms like *Nitzschia gracilis*, *Syndera rumpens*, *S. delicatissima* are exposed to acidity to a greater degree due to their larger surface area, whereas smaller diatoms (e.g., *Achnathes minutissima*, *Coccneis placentula*, *Nitzschia fonticola*, and *N. lacuum)* are presumably less exposed to acidity and associated aluminum (Genter and Amyot 1994). Phytoplankton associated with acidic wetlands include the dinoflagellates, *Peridinium inconspicuum* and *P. limbatum* (Baker and Christensen 1991). Under acidic conditions, *Merismopedia*, *Peridinum*, and *Gymnodinium* are common species (Dixit and Smol 1989). Also, as acidification progresses, planktonic algae can become less dominant and benthic algal species can increase (e.g., , *Mougeotia* sp. and *Spirogyra* sp.; Dixit and Smol 1989).

Diatoms in particular are sensitive indicators of acidity, as shown by studies where they successfully predicted acidity of New York lakes (Dixit *et al.* 1992, Dixit et al. 1993, Dixit et al. 1999), southeastern Ontario lakes (Christie & Smol 1993), Montana lakes (Charles *et al.* 1996), Montana wetlands (Apfelbeck 1998), streams in the mid-Atlantic Highlands ecoregion (Pan *et al.* 1996), Chesapeake Bay waters (Cooper 1995), and in other regions (e.g., Birks *et al.* 1990, Anderson et al. 1993, Battarbee et al. 1999). Although pH appears to a be an important factor in determining the diatom composition, other factors such as dissolved oxygen concentrations, nutrient loading, metal and micronutrient availability, and lake size and shape are also important (Cook and Jager 1991).

Effects on Biomass, Growth

Algal biomass in a stream declined as pH dropped from 6.5 to 4.5, and the community composition shifted to larger cells that were less productive (Havens 1992). The combination of acid and aluminum is much more harmful to growth of diatoms and cyanobacteria than acid alone. Growth of *Schizohrix calcicola*, a filamentous cyanobacteria, was inhibited by all aluminum treatments but not inhibited by acidic conditions (Genter and Amyot 1994). Some species, such as *Rhopalodia gibba* and *Synedra rumpens*, tend to decrease in abundance with more acidic conditions but do not decrease further with the addition of aluminum (Planas *et al.* 1989). Filamentous cyanobacteria (e.g., *Schizohrix calcicola)* increase under acidic conditions but decrease under a combination of elevated acidity and aluminum (Dixit *et al.* 1991b).

3.5 Effects of Salinization

Algal species are sensitive indicators of salinity and conductivity in surface waters and sediments of lakes and wetlands. Salinity ranges associated with the occurrence and greatest density of particular species are described by Blinn (1993) and Fritz *et al.* (1993). Studies of saline lakes in western North America found that the greatest richness of diatom taxa occured with specific conductance of less than 45 mS (Blinn 1993), although light, temperature, and nutrient levels may influence response to salinity in prairie wetlands (Robarts et al. 1992). Among some lacustrine wetlands in Wyoming, relative production of different algal types (epiphyton, epipelon, phytoplankton) varied depending on salinity and associated floating macrophytes (*Chara* at low salinity, *Potomogeton* at higher), but total algal production did not. At lower salinity, epiphyton predominated whereas at higher salinity phytoplankton and benthic algae assumed a larger proportion of the total production (Hart & Lovvorn 2000). In western Kentucky wetlands that received acid mine drainage, conductivity -- not pH -- was a stronger determinant of diatom assemblages (Pan & Stevenson 1996).

3.6 Effects of Temperature

Apparently no studies have been published in the last decade regarding response of algae to temperature changes in wetlands. Algal biomass generally increases with rising temperature and duration of growing season.

3.7 Effects of Sedimentation, Burial

The impacts of sedimentation on algae have not been documented within the last decade in wetlands. Attached algae such as periphyton and benthic algae are probably affected disproportiately.

3.8 Effects of Vegetation Removal

The removal of wetland vascular plants will obviously cause a decrease in the abundance of algae associated with these plants, i.e., epiphytic algae (Murkin *et al.* (1994). Epiphyton biomass varies by vascular plant species. For example, epiphyton biomass was found to be higher on *Polygonum* sp. than on *Typha* sp. (Cronk and Mitsch 1994a, b). Vegetation removal also can have a cascading trophic effect. In a pond experiment in California, the removal of vegatation resulted in growth of predaceous beetles which reduced chironomid populations. This reduction in grazers allowed the periphyton population to grow independently of herbivore pressures (Batzer and Resh 1991).

3.9 Effects of Turbidity, Shade

Shade tolerance differs among algal species, allowing some species to thrive at the expense of others (Steinman and McIntire 1990, Steinman et al. 1990). In the Florida Everglades, denser stands of vascular plants had less epiphytic algae due to growth limitation by shade, and consequently removed less phosphorus (but more nitrate) than more open stands (Grimshaw *et al.* 1997). Algal responses to nutrient additions can be confounded by shading (Stevenson et al. 1991). Tannins produced by some wetland plants under acidic conditions can also stain waters to such a degree that light transmissivity is reduced, with consequent reductions in algal growth and changes in species composition (Graham 1989, Goldsborough and Brown 1991).

Turbidity-related shading probably impacts benthic algae more than phytoplankton (Mitsch & Reeder 1991). In a Canadian marsh, shade from metaphyton reduced growth of submerged macrophytes and presumably their associated epiphytic algae; the metaphyton had expanded in response to nutrient additions (McDougal *et al.* 1997). Despite the shading out of some species, primary production increased two-fold due to the large amount of metaphyton (Hann and Goldsborough 1997). Shade from high densities of nannoplankton, such as occur when planktivorous fish are present , can also reduce the amount of benthic and epiphytic algae (Mazumder *et al.* 1989). Some prostrate diatoms, such as *Achnanthes lanceolata* and *Cocconeis placentula*, appear to gain competitive advantage under conditions of low light (Steinman and McIntire 1990). In shallow lakes in which fish increased turbidity by resuspending sediment, nanochlorophytes replaced centric diatoms and larger diatoms replaced smaller types (Havens 1991a).

3.10 Effects of Dehydration, Inundation

Processes

Dehydration and inundation affect algal communities by altering the available water volume (Hough *et al.* 1991), substrate area (Sand-Jensen and Borum 1991), wind mixing (Robinson *et al.*1997b), scouring (Biggs and Close 1989, Stevenson 1990), temperature, and nutrient concentrations. Flooding also can alter algal communities by dispersing algae and by increasing turbidity, thus decreasing light penetration. Water level drawdown can concentrate nutrients, contaminants, and salinity. However, nutrient availability can increase with flooding as well as with drought (Cronk and Mitsch 1994). Thus, algal response to nutrients in shallowly- or temporarily-flooded wetlands is more complex than in lakes (Robarts et al. 1995, Moss *et al.* 1996).

Some algae have adapted to periodic dessication by achieving the ability to produce thick cell walls, mucilage sheaths, zygospores, or cysts (Steinman and McIntire 1990). Taxa that experience frequent drawdowns, such as those in temporary wetlands, are most likely to have such adaptations, whereas taxa that are normally submerged year-round are more susceptible to dessication. Some algal taxa found in soils can survive dessication for over 30 years (Trainor and Gladyeh 1995). Moderate currents and flooding (e.g., 41 cm of water depth per week) can provide epiphytic algae with nutrients from outside a wetland while removing wastes, thereby increasing growth (Cronk and Mitsch 1994).

Effects on Species Composition

The conversion of shallowly-inundated wetlands to deepwater ponds or lakes reduces epiphyton and metaphyton, and causes algal communities to shift toward a greater dominance by phytoplankton (Robinson *et al.* 1997a,b). This can trigger a shift in the invertebrate community, from detritivores and/or scrapers to filter-feeding taxa. In intermittent bay wetlands in South Carolina and Georgia, paleolimnological analysis indicated that *Eunotia* spp., *Luticola saxophila*, and *Pinnularia borealis* var. *scalaris* were indicators of sustained drying or draw-down; and other *Pinnularia* spp. and *Stenopterobia densestriata* responded positively to persistent ponding (Gaiser et al. 1998).

Effects on Density, Biomass, Growth

Initially, an increase in water levels in a Manitoba marsh (a 30 to 60 cm total depth increase) increased the biomass of floating metaphyton, epiphytes, epipelon and phytoplankton. However, the algal response did not persist (Hosseini and van der Valk 1989a, Robinson *et al.*1997a).

3.11 Effects of Other Human Influences

Humans can indirectly alter the species composition of algae by removing or introducing grazing and planktivorous fish (Hanson and Butler 1990), but this effect is not inevitable (e.g., Ramcharan *et al.* 1995). Algae are also affected when introduction or harvesting of fish, or activities or events, change the density of wetland invertebrates, because grazing invertebrates significantly reduce algal biomass in wetlands (Hann 1991, Botts 1993). In an experimental pond system, grazing from macroinvertebrates resulted in a simplified algal community dominated by *Stigeoclonium* sp. (Hann 1991). Grazing by invertebrates causes a disproportionate reduction in more edible algae species, allowing less palatable species (e.g., blue-greens) to prosper (Goldsborough and Robinson 1996, Hann and Goldsborough 1997).

Algal communities also can be physically impacted by waves and scouring. Older colonies (e.g., 12, 18 and 24 days old) are more resistant to scouring than younger (e.g., 6 day old) communities, which suffered a 47.6% reduction in biomass following disturbance by waves (Peterson *et al.* 1990). Resistance to wave scouring was due to stabilization of algal mats by diatom mucilages and overlying *Oscillatoria* sp. surface layers. Recovery of communities was related to differential settling rates of algal species. *Fragilaria* sp. and *Navicula* sp. resettled quickly while *Nitzschia* sp. were not replaced as readily.

3.12 Wetland Monitoring

Spatial and Temporal Variation

In rivers, algae sampled using artificial substrates appear to integrate the river environment fairly well. For example, in one river system the assemblage of algal species that colonized artificial substrates was remarkably similar among samples collected from 1st through 5th order channels (Molloy 1992). However, in wetlands the density and species composition of algal communities often varies over the scale of a few meters horizontally and a few centimeters vertically. These fine-scale spatial patterns, which confound attempts to characterize entire wetlands or even particular habitats within wetlands (Morin and Cattaneo 1992), are sometimes driven by spatial patterns in nutrient inputs (e.g., Wu and Mitsch 1998, McCormick 1998) or physical habitat structure (Rose and Crumpton 1996).

Temporal variation is also important, at both a seasonal and annual scale (Whitton et al. 1998). In a riverine study of 186 algal species, the possible effects of water quality were subsumed by other differences between years (Miller *et al.* 1992). Within years, many water bodies have predictable, seasonal shifts in the community structure and spatial distribution of phytoplankton and other types of algae (Harper 1992, Cloern et al. 1992), and these may be altered by nutrient additions (Gabor et al. 1994). Diatoms are often more common in the spring and summer, green algae in the summer, and cyanobacteria in late summer (Harper 1992). Seasonal variation in the relative abundance of algae species can be dampened by eutrophication (McCormick *et al.* 1997). Oligotrophic Everglades marshes sampled during wet and dry seasons exhibited seasonal variation in diversity, with cyanobacteria (e.g., *Chroococcus turgidus*, *Scytonema hofmannii*) dominant during the wet season, and diatoms (e.g. *Amphora lineolata*, *Mastogloia smithii*) during the dry season. Eutrophic marshes that were dominated by Cyanobacteria (e.g., *Oscillatoria princeps*) and green algae (e.g., *Spirogyra* sp.) exhibited comparatively little seasonality.

Equipment and Techniques

Algae are relatively simple to collect and the EPA and others have developed standardized sampling protocols for streams, rivers, and lakes (Stevenson and Bahls 1999): http://www.epa.gov/owow/monitoring/rbp/ch06main.html .

These protocols will need to be adapted for use in wetlands, as the equipment designed for other systems is often hindered by wetland conditions. A variety of equipment and techniques are used in monitoring algal communities in wetlands, and are summarized by Aloi (1990), Adamus and Brandt (1990), and Adamus (1996). Artificial substrates in the form of glass “diatometer” slides (McCormick *et al.* 1996), polyvinyl substrate (Mazumder *et al.* 1989), open cell styrofoam (Bothwell 1989), acrylic rods (Hann 1991), and nutrient-diffusing alginate (Gensemer 1991) have been used, but not without controversy (Aloi 1990). Colonization of natural wetland substrates also has been monitored (Batzer and Resh 1991). Metaphyton have been sampled by removing of a portion of numerous mats and combining them into one sample to mediate the effects of spatial variation (McCormick and O’Dell 1996). Corers are sometimes used to sample benthic algae (Glew 1991). Phytoplankton are typically sampled with water bottles or other volumetric samplers.

Measuring algal biomass, volume, and density (e.g., McCormick et al. 1998, Hillebrand et al. 1999) with high precision is time-consuming and probably unnecessary for most assessments of wetland condition. Formulas have been devised and tested for relating algal volume in wetlands to simpler measurements of chlorophyll-a (LaBaugh 1995). Measuring algal metabolic activity also may be useful for some objectives, but requires repetitive measurements of changes in dissolved oxygen (DO) concentrations or net uptake of radiolabelled CO_2 added to water (Wetzel and Likens 1991, Keough et al. 1993, McCormick et al. 1997). Numerous published estimates of algal production in freshwater and saltwater wetlands were compiled by Goldsborough and Robinson (1996).

Analysis of diatom remains in lake and wetland sediment cores continues to be a promising but painstaking method for establishing water quality reference (presettlement) conditions, as discussed by Battarbee *et al.* (1990), Van Dam and Mertens (1993), and Dixit et al. (1999).

Identification of Taxa

Some of the more recent references for identifying North American algae include Cox (1996) for diatoms and Dillard (1989a, b, 1990, 1991, 1993, 1998, 2000) for the southeastern United States. Opinions are mixed regarding the necessity of identifying algal taxa to species or genus. While finer-level identification is generally desirable, this decision clearly depends on the available resources and the objective of the study. Identification of algae to only the genus level or coarser can produce information useful for some objectives (e.g., Prygiel and Coste 1993).

Metrics for Assessing Impacts to Wetland Algal Communities

Metrics used for characterizing algal community response to pollution in streams are described by Stevenson and Bahls (1999) (http://www.epa.gov/owow/monitoring/rbp/ch06main.html). By grouping species according to reputed tolerances, much of the statistical variation inherent in examining single species can be compensated for (McCormick et al. 2000). Use of weighted average regression analysis can elucidate relationships with specific causative factors, e.g., nutrients (Hall & Smol 1992, Line et al. 1994, Lowe and Pan 1996, Stevenson et al. 1999, Winter & Duthie 2000).

The effectiveness of 5 indices based on diatoms, that are used to assess condition of European waters, was evaluated by Kelly *et al.* (1995). In North America, models using algae to predict total phosphorus and conductivity were developed and tested by Pan and Stevenson (1996). In general, their model that used planktonic algae was better at predicting conductivity than models based on epiphytic diatoms. In contrast, the model using epiphytic diatoms was better at predicting total phosphorus than models based on planktonic diatoms.

In a study of 356 Montana lakes, reservoirs, and wetlands, algal species percent dominance and community diversity were not successful in reliably differentiating habitats, whereas pH and conductivity provided a firmer classification (Charles *et al.* 1996). The usefulness of diatoms (Dixit *et al.* 1992, Stoermer & Smol 1999, Slate & Stevenson 2000) and benthic algae (Whitton *et al.* 1991, Lowe and Pan 1996) for detecting long-term environmental change has been described in various types of wetlands. However, in prairie wetlands, Mayer and Galatowitsch (1999) found diatoms to be a poor indicator of human-related disturbance. This was attributed to the high degree of natural variability in the sampled wetlands, which was inadequately addressed by the wetland classification that was employed in sampling. This prohibited statistical separation of the relative influences of human-related and natural disturbance.

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Section 4: Vascular Plants

4.1 Use as Indicators

This section addresses woody and herbaceous plants that grow in non-tidal wetlands. For a general discussion of the topic based on pre-1990 scientific information, and for discussion of advantages and disadvantages of using plants as indicators of wetland integrity, readers should refer to Adamus and Brandt (1990). Also, some of the recent literature on use of wetland plants as indicators of water quality is summarized, especially for Europe, by Doust et al. (1994).

Vascular plants, also termed macrophytes, are commonly classified as woody or herbaceous. Woody plants may be classified further as trees, shrubs, or vines. Herbaceous plants (which in practice sometimes include non-vascular mosses and ferns) may be classified as submersed, floating-leaved, or emergent. Plants are such obvious components of wetlands, and they are so sensitive to wetland hydrology, that they are commonly used to delineating wetland boundaries. A good background on the ecology of wetland vegetation is provided by Keddy (2000) and Cronk and Fennessy (2001). Information on plants in riparian systems and floodplain wetlands is compiled and summarized by Malanson (1993), Galatowitsch and McAdams (1994), and Patten (1998).

Wetland plants, and especially herbaceous species, are increasingly being used as indicators of wetland ecological condition in North America. The 1990's have seen increased use of plants as indicators of wetland ecological condition. Following the lead of European botanists (e.g., Robach et al. 1996), wetland botanists in North America have increasingly attempted to identify patterns of plant community response to human-related alteration of streams (Small et al. 1996) and wetlands (Wilcox 1991). Recently, attempts have been made to develop and/or test multimetric indices for the purpose of estimating condition of wetlands, e.g., in North Dakota, Minnesota, Ohio, Pennsylvania, Delaware, Oregon, and perhaps elsewhere. These indices are often termed "floristic indices" (e.g., Andreas & Lichvar 1995). As a partial starting point for such indices, Adamus and Gonyaw (2000) compiled literature and prepared a documented species database for EPA's internet web site, that categorizes many wetland species as tolerant or intolerant, with regard to overall sensitivity, and/or specific sensitivity to excessive nutrients and hydrologic alteration: http://www.epa.gov/owow/wetlands/bawwg/publicat.html

In the past decade, several studies have used plant assemblages specifically to indicate the ecological condition of a large series of wetlands. Results have been published from such studies, for example, in the Seattle area (Cooke & Azous 2000), Massachusetts (Carlisle et al. 1998), Montana (Apfelbeck 1998), Minnesota (Gernes and Helgen 1999), and western Oregon (Magee et al. 1999, Adamus 2001). Most of these studies are detailed at: http://www.epa.gov/owow/wetlands/bawwg/case.html .

Also, in Minnesota wetland plants were used to represent the condition of landscapes that contained a large wetland/ riparian component (Galatowitsch et al. 1998, Mensing et al. 1998). Sampling 15 wetlands belonging to each of 8 wetland types, the investigators found positive or negative correlations of the several metrics with a site disturbance score and/or various land cover types measured within 500, 1000, and 2500 m of each wetland. Results are reported by wetland type at: http://www.hort.agri.umn.edu/mnwet/

4.2 Effects of Enrichment/ Eutrophication/ Reduced Dissolved Oxygen

Processes

Excessive nutrients can affect wetland plant communities in a variety of ways (Wisheu et al. 1990, Weisner 1990), including:

> (a) shifting the species composition away from species that take up nutrients slowly, to those that are able to exploit nutrient pulses more rapidly or which have high nutrient requirements (Hough *et al.* 1989, Arts *et al.* 1990, Gopal and Chamanlal 1991, Wetzel and van der Valk 1998);
>
> (b) triggering algal blooms that can shade out many submersed herbaceous plants (Mason 1990, Crowder and Painter 1991, Stevenson *et al.* 1993, Srivastava *et al.* 1995, Short and Burdick 1995, Coops and Doef 1996);
>
> (c) causing dead plant material to accumulate faster than it can decompose completely, thus altering understory and soil structure (Neill 1990, Craft & Richardson 1993).

Over the long term, nutrient additions to most wetlands tend to reduce species richness and increase the dominance of a few species. Often, non-native species are most capable of invading rapidly changing environments. Consequently they frequently come to dominate some enriched wetlands.

Woody plants do not usually show an immediate obvious visible response to nutrients. Nutrient-related shifts in community composition, if they occur at all, do so over long time periods. However, seed germination, seed production, foliar nutrient concentrations, shoot length, and annual growth of woody plants are all potentially affected by enrichment and can be measured to document short-term nutrient exposure. In relatively unaltered watersheds, floodplain trees farthest from the channel may suffer the least from nutrient deficiencies, due to less frequent flooding, whereas in watersheds with largely agricultural land cover, near-channel trees may be less prone to nutrient limitation (Friedman et al. 1996).

Response of wetland plants to nutrient additions can be influenced by many factors, including type of nutrient, dosing rate, nutrient concentration, soil or water pH, hydrologic conditions, season, plant species, and life stage. Together, these factors may determine whether a wetland plant community is limited more by a lack of nitrogen, phosphorus, potassium, calcium, or other elements (e.g., Craft et al. 1995). Bog vegetation is often limited the most by lack of nitrogen (Bridgham et al. 1996), whereas vegetation in tidal freshwater wetlands may be limited more by phosphorus.

Species Richness

Enrichment can increase or decrease species richness of plants within a wetland, depending on the initial species mix, nutrient loading rates, season, and other factors. In France, wet meadows receiving agricultural fertilizer (30 to 160 kg N/ha) had lower species richness, and their dominant species were common in other wetlands (Grevilliot *et al.* 1998).

Species Composition

Because the database on EPA's Biological Assessment of Wetlands Working Group (BAWWG) web site (Adamus and Gonyaw 2000) compiles the literature on nutrient-related species composition shifts, we have limited the discussion of autecological sensitivities in this document.

Many wetlands are naturally eutrophic. Plants that typify such situations often include species such as *Typha* sp., *Phragmites* sp., *Lythrum salicaria,* and *Lemna* (Grace 1989, Davis 1991, Huebert and Shay 1991, Davis 1994, Otto et al. 1995, Weiher et al. 1996, Doren *et al.* 1997) which are relatively invasive and tend to form monotypic stands. Some species occur only in eutrophic or oligotrophic waters, whereas others span the full range of nutrient conditions (e.g., Srivastava *et al.* 1995). Among the species that characterize nutrient-rich wetlands, some dominate not because they *require* high nutrient levels, but because they have life history characteristics that allow them to invade and spread rapidly into enriched habitats where other species are or would be struggling. Nutrient limitation of Florida Everglades plants was explored by Daoust and Childers (1999). They found wet prairies were highly P-limited at N:P ratios above 36:1 and *Cladium jamaincense* remained dominant, with sub-dominants including *Peltandra virginica*, *Pontedaria cordata*, *Saggitaria lancifolia* and *Panicum hemitomon*. When N:P ratios dropped below this threshold, *Typha* spp. became increasingly dominant. *Hymenocallis palmeri* was shown to be N-limited and may signal a change in nutrient regime.

Bogs and other nutrient-poor wetlands are logically the most sensitive to nutrient additions (Moore *et al.* 1989). There, the increased availability of nutrients allows grasses and common opportunistic plants to out-compete the rare, nutrient-poor specialists such as sundews, orchids, and pitcher plants. However, plant community response to enrichment of boreal wetlands depends largely on the species present (Nams *et al.* 1993).

Nutrient additions to wetlands do not inevitably cause shifts in species composition, at least not in the short term, in wetlands with just a few dominant (and perhaps "adapted") species. When nutrients were added to an Everglades fen, areas with moderate (500-750 mg/kg) and low (<500 mg/kg) phosphorus soil concentrations maintained their original plant composition over many years (Richardson *et al.* 1999).

Impacts of enrichment are often confounded or even obscured by simultaneous effects of correlated disturbances. In a southeastern Michigan lake (Hough *et al.* 1991), a combination of water level and nutrient declines resulted in a shift from a *Potamogeton*-dominated community to one dominated by *Nuphar*, *Nymphaea* and *Myriophyllum.* In the Everglades, shifts in plant growth and species composition in response to nutrients depend partly on water levels and/or fire history (Grace 1989, Davis 1989, Davis 1991, Urban *et al.* 1993, Maceina 1994, Craft *et al.* 1995, Newman *et al.* 1996, Kludze & Delaune 1996, David 1996, Doren *et al.* 1997).

Because enrichment can increase densities of phytoplankton and epiphytes, and this reduces the amount of light available to submersed plants, eutrophic lakes are often dominated by phytoplankton and non-rooted macrophytes, whereas oligotrophic lakes are dominated by rooted macrophytes and a higher proportion of submersed plants (Hough *et al.* 1989, Srivastava *et al.* 1995). When nutrient additions are curtailed, submersed macrophytes may increase and consequently stabilize sediments (Scheffer 1991, Stevenson *et al.* 1993).

Density, Biomass, Growth, Productivity, Germination

Many aquatic plant species respond to nutrient additions by increasing their growth, biomass, and productivity. *Growth* responses to enrichment have been documented for only about 80 wetland-associated species in North America, and of these, most have tolerated enrichment or responded to enrichment with increased biomass or growth (Adamus & Gonyaw 2000). Wetland macrophytes may be more nutrient-limited than are algae (Duarte 1992). In nutrient limited systems, growth increases can be dramatic. Fertilization of an Alaskan river resulted in an increase in bryophyte biomass from 17g dry mass/m^2 to 322 g dry mass/m^2 (Bowden *et al.* 1994). Fertilization of the common bog plant *Calluna vulgaris* with a garden fertilizer led to increased flowering per shoot and a greater proportion of flowering shoots (Iason and Hester 1993). In alpine wet meadows of Colorado, fertilization increased the biomass of grasses at the expense of forbs (Bowman et al. 1993). Overall nutrient uptake, photosynthesis, and growth of the dominant species were not strongly affected by application of 500-1000 g N and/or P to each of five 2 x 2 m plots.

The response of plant growth to nutrient addition can vary depending on the degree to which the species that are present allocate nutrients to roots *vs.* shoots, as well as their overall life history strategies (Carter and Grace 1990, Grace 1990, Keddy 1990, Olff *et al.* 1990). Many plant species allocate more biomass to shoots during competition for light and more to roots when competing for nutrients (Tilman 1990, 1991, Poorter and Remkes 1990, Poorter and Lambers 1991). For example, *Phalaris arundinacea* and *Echinochloa crusgalli* have greater root to shoot biomass ratios under lower nutrient levels (Figiel *et al.* 1995). *Potamogeton nodosus* responds to nutrient additions by allocating biomass predominantly to tubers while *Hydrilla verticillata* concentrates nutrients in aboveground structures (McFarland *et al.* 1992).

Increases in aboveground biomass can smother other plants following senescence of tissue, thus helping maintain the dominance of species that exploit nutrients the most (van Auken and Bush 1992). When nitrogen and phosphorus fertilizers were added to a *Typha glauca* and a *Scolochloa festucacea* marsh during two growing seasons, this resulted in increased biomass of both *Scolochloa festucacea* and *Typha glauca*. However, biomass of *Scolochloa festucacea* declined in the second year due to large accumulated amounts of *Typha glauca* litter (Neill 1990).

Fertilizer applications of up to 90 kg/ha can increase the aboveground productivity of sedge-dominated wet meadows, but applications higher than that rate had little added effect (Reece *et al.* 1994). Submersed macrophytes in non-eutrophic waters generally increase in response to moderate nutrient additions, even though most derive their nutrients from sediments rather than the water column (Spencer 1990, McFarland and Barko 1990, Barko et al. 1991, Spencer *et al.* 1993, Spencer and Ksander 1995).

Carbon dioxide is another type of nutrient utilized by wetland plants. During a laboratory experiment *Callitriche* was unable to grow under ambient conditions, but CO_2 enrichment with 500-1000 uM of carbon dioxide led to growth of 0.089 to 0.124/day. Under the same concentrations *Elodea* growth rates increased from 0.046-0.080 to 0.076-0.117/day (Vadstrup and Madsen 1995).

Enrichment also affects germination rates of some macrophytes. *Typha latifolia* germinated in fewer days than did *Typha domingensis* at high phosphate levels (.200 mg/L). However, the germination rate of each species was unaffected by other nutrient levels (Stewart *et al. 1*97). Phosphorus amendments led to improved establishment of *Sphagnum* and *Eriophorum angustifolium* in laboratory experiments (Ferland and Rochefort 1997).

Cottonwood trees treated with 50 kg/ha fertilizer increased stem volume four-fold (van den Driessche 1999). In Appalachian peat bogs, spatial dominance of bristly dewberry, *Rubus hispidus,* was positively related to nutrient levels, but dominance of the Ericaceae shrubs was negatively related (Stewart and Nilsen 1993).

4.3 Effects of Contaminant Toxicity

Much of the literature on aquatic plant sensitivity to chemical contaminants, as revealed by experimental dosing (phytotoxicity testing), is summarized by Lewis (1995). In general, dose-response relationships are less well known for vascular plants than for algae, and most experiments have used floating-leaved plants (especially duckweed, *Lemna* spp.) rather than rooted plants.

Processes and Symptoms

Most vascular plants are relatively tolerant of contaminant toxicity. When effects occur, they usually result from the effects of contaminants on plant metabolic pathways, enzymatic reactions, and growth (Fitter and Hay 1987). Symptoms of toxicity can include growth reduction, small leaves, necrotic, chloritic or discolored leaves, early leaf fall, stunted root growth, suppressed growth of lateral roots and death of root meristems (Pahlsson 1989, Rhoads *et al.* 1989, Vasquez *et al.* 1989, Alloway 1990, Kiekens 1990, O'Neill 1990, Kabata-Pendias and Pendias 1992, Dushenko *et al.* 1995). Acidic conditions in some wetlands can increase the harmful effects of many heavy metals (e.g., Carlson and Carlson 1994).

Effects on Species Composition

Shifts in wetland plant species composition in response to contaminants have not been widely documented. Thus, the ability of plant-based multimetric indices to represent chemical contamination of wetlands is, at best, currently very limited. Submersed species tend to accumulate contaminants and also are perhaps the most sensitive plants to contaminants (Outridge and Noller 1991). For example, in acidic lakes of New Jersey, submersed species contained higher trace element levels than did floating-leafed species, although one floating-leaved species (water shield, *Brasenia*) bioaccumulated zinc and cadmium substantially

(Sprenger and McIntosh 1989). In an Ontario lake, cattail stands appeared to tolerate acid mine drainage and associated heavy metals. Toxicity of acid mine drainage to macrophytes often depends on local environmental and geological features that alter contaminant bioavailability (Fyson *et al.* 1991). Differences also exist among species with regard to their sensitivity to particular herbicides.

Effects on Biomass, Growth, Health

The invasive submersed plant, *Myriophyllum spicatum*, did not grow when exposed to cadmium concentrations above 7.63 μg /mL (Sajwan and Ornes 1996). Arsenic, cadmium, copper, lead, and zinc inhibited growth in hybrid poplar (*Populus*) and several other tree species (Lejeune *et al.* 1996). Iron and manganese, although not usually toxic to wetland plants, do affect species in some wetland types. For example, laboratory experiments revealed differences among 44 fen species with regard to the influence of iron on growth (Table 4.1) (Snowden and Wheeler 1993). Boron can also be toxic. When added at rates of 0, 2, 4, 8, and 16 mg/L in laboratory experiments, it caused significant decreases in the growth of seedlings of several wetland trees: *Betula nigra*, *Nyssa aquatica*, *Platanus occidentalis* and *Taxodium distichum.* Severe leaf damage, but no reduction in growth, occurred at the higher boron levels in *Quercus alba*, *Quercus falcata* var. *pagodaefolia*, *Quercus nigra*, *Quercus michauxii* and *Quercus phellos* (McLeod and Ciravolo 1998). In a lacustrine wetland exposed to high arsenic levels, cattails (*Typha latifolia*) were shorter and had necrosis of leaf tips and reduced micronutrient concentrations in root tissues. These symptoms were observed at sediment and water concentrations exceeding 300 μm/g and 400 μm/g arsenic respectively (Dushenko *et al.* 1995).

However, another study found that seedlings of *Typha latifolia* were able to tolerate and accumulate zinc (1.0 μg/ml), lead (10.0 μg/ml) and cadmium (0.2 μg/ml) (Ye *et al.* 1998). Laboratory experiments indicated that *Eriocaulon septangulare* was unaffected by tissue cadmium concentrations of less than 2.6 μg/g dry weight in shoots and less than 45 μg/g in (Stewart and Malley 1999). The emergent herbs *Bacopa monnieri* and *Scirpus lacustris* were tolerant of cadmium and copper additions of up to 5 μM/mL, although decreases in chlorophyll concentration occurred (Gupta *et al.* 1994).

Oil spills can have long-lasting effects on wetland plant communities (Obot et al. 1992). In a greenhouse experiment, oil and a detergent used to clean up oil spills were applied to *Sagittaria lancifolia*, *Scirpus olneyi* and *Typha latifolia.* The leaves on all of the study plants died following oiling, but new leaves soon developed on those plants subjected to oil and subsequent cleaning with the detergent. *Scirpus olneyi* was the least sensitive of the three species whereas *Typha latifolia* appeared to be the most sensitive (Pezeshki *et al.* 1998).

Nitrile and volatile organic acids in culture media were very toxic to *Azolla filiculoides* plants. In secondary effluent containing nitrile, the plants shed their roots, suffered fragmentation of their fronds, and eventually died (Kitoh *et al.* 1993).

The herbicides, Rodeo™ and Garlan 3A™, applied to control *Lythrum salicaria*, also decreased growth rates of non-target species such as *Lemna gibba* (Gardner and Grue 1996). The herbicide

Triclopyr™ has been reported to be relatively harmless to wetland vascular plants (Gabor *et al.* 1993).

The germination of the forbs *Echinochloa crusgalli* and *Sesbania macrocarpa* was unaffected by unspecified pollutants from a coke plant, pulp mill, and a wastewater treatment plant. Seedling growth of *Echinochloa crusgalli* increased after exposure to pollutants from one wastewater treatment plant but not from another (Walsh *et al.* 1991).

Bioaccumulation

Arsenic, cadmium, copper, lead, aluminum, and zinc often accumulate in plants growing near industrial areas (St-Cyr & Campbell 1994) and mining sites, frequently at levels toxic to other ecosystem components and the plants themselves (O'Niell 1990, Kabata-Pendias and Pendias 1992, Ton et al. 1993). In Montana, upland soils with high levels of arsenic and other metals from smelter emissions had reduced cover and vertical diversity of plants, lower species richness, and increased dominance of weedy species (Galbraith et al. 1995). In Colorado, riparian conifers and *Populus tremuloides* died when exposed to high levels of iron and manganese, whereas *Populus angustifolia* and *Salix monticola* remained healthy (Barrick and Noble 1993). In Florida, constructed marshes effectively removed methyl-mercury from Everglades water (Miles and Fink (1998).

Plants do not inevitably bioaccumulate or biomagnify metals from sediments. Moreover, when they do accumulate, effects are not always obvious. In marshes of northern Canada, no correlation was found between spent gunshot and lead in soil and lead in plant tissue, which remained at background concentrations (Tsuji and Karagatzides 1998). In a southeastern wetland, the dominant tree, *Pinus taeda*, accumulated metals and other trace elements, but high levels of metals in the soil did not prevent growth of seedlings (Carlson and Carlson 1994). Extensive spatial and species variation in tendency of plants to accumulate contaminants has been noted in Lake Ontario marshes (Crowder and Painter 1991).

Much of the literature pertaining to plant bioaccumulation of heavy metals was reviewed by Crowder (1991), Nellessen & Fletcher (1993), and Odum et al. (2000). Plant uptake of metals appears to be largely influenced by pH, Eh, soil organic content, oxide and carbonate content, and cation exchange capacity (Crowder 1991), as well as plant species characteristics (Jackson & Kalff 1993, Thompson et al. 1999). Shallow-rooted marsh species may be more effective sinks for lead than are woody plants (Ton et al. 1993). Wetland plants can also take up some toxic hydrocarbons (Gobas et al. 1991).

Table 4.1. Tolerance groupings of fen plant species exposed to various iron concentrations for a two week period (Snowden and Wheeler 1993).

INSENSITIVE	SLIGHTY SENSITVE
Juncus articulatus	*Valeriana dioica*
Eriophorum angustifolium	*Holcus lanatus*
Carex echinata	*Juncus subnodulosus*
Juncus effusus	*Phalaris arundinacea*
Juncus acutiflorus	*Lysimachia vulgaris*
Iris pseudacorus	*Carex lepidocarpa*

Molinia caerulea
Juncus articulatus
Oryza sativa
Agrostis stolonifera
Eriophorum latifolium
Pedicularis palustris
Parnissia palustris
Carex pulicaris

Juncus inflexus
Potentilla palustris
Ranunculus flammula
Briza media
Carex diandra
Caltha palustris
Phragmites australis

MODERATELY SENSITIVE
Galium palustre
Carex appropinquata
Lotus uliginosus
Trifolium pratense
Primula farinosa
Epilobium palustre
Thalictrum flavum
Galium aparine
Eupatorium cannabinum
Valeriana officinalis

VERY SENSITIVE
Filipendula ulmaria
Lychnis flos-cuculi
Rumex acetosa
Scrophularia auriculata
Rumex hydrolapathum
Epilobium hirsutum

4.4 Effects of Acidification

Processes

Acidic conditions in wetland soils increase the toxicity of aluminum and manganese (Rendig and Taylor 1989, Crowder 1991). Acidification can directly impact plants by limiting the availability of some inorganic nutrients and carbon (Farmer 1990). Acidic conditions also promote the conversion of nitrates into ammonium. In regions with carbonate soils, acidification can mobilize phosphorus (Reddy et al. 1993). Acidic conditions also can impact plants indirectly by reducing densities of grazing and detritus-processing invertebrates. Mosses can alter the pH of water, increasing the bioavailability and toxicity of some metals to other wetland species, and thus potentially causing shifts in species composition (Vitt & Chee 1990, Vedagiri & Ehrenfeld 1991, Kooijman and Bakker 1994).

Effects on Species Composition and Richness

Effects of acidification (or its reversal by liming) on wetland species composition are not consistent among wetland types or even within individual wetlands (Farmer 1990, Baker and Christensen 1991, Mackun *et al.* 1994, Weiher et al. 1994). Many plant species that inhabit bogs and pocosin wetlands are, of course, adapted to tolerating acidity levels that would kill most wetland plant species. Species whose decline or disappearance from a lake coincided with acidification include *Lobelia dortmanna*, *Isoetes riparia*, *Myriophyllum tenellum*, *Nuphar* sp., *Utricularia vulgaris*, and *Potamogeton epihydris* (Farmer 1990). Species whose relative abundance increased include *Leptodictium riparium*, *Eleocharis acicularis*, *Sphagnum* sp., and *Eriocaulon septangulare* (Farmer 1990). Within a two year period after 1100 mg of lime was applied to a 100 hectare Adirondack watershed, only 6 of 64 wetland plant taxa increased in cover, frequency, or importance. The cover of *Sphagnum* and *Carex interior* decreased in control plots compared to limed plots. Cover of *Cladium mariscoides* tripled in extent. In contrast, *Drosera intermedia, Hypericum canadense,* and *Muhlenbergia uniflora* all were negatively impacted by the liming.

Among 51 Maine peatlands, plant species richness tended to increase with decreasing acidity and rising pH (Anderson & Davis 1997), as was also the case in Minnesota peatlands (Glaser et al. 1990) and northern Ontario wetlands (Jeglum et al. 1995). Species associated with very or moderately acidic conditions included *Smilacina trifolia, Carex oligosperma, Chamaecyparis thyoides, Pinus strobus, Utricularia cornuta, Vaccinium angustifolium, V. oxycoccus, Gaylussacia dumosa,* and *Kalmia angustifolia.* Species associated with less acidic conditions in the Maine peatlands included *Alnus rugosa, Campylium stellatum, Trichophorum alpinum, Rhamnus alnifolia, Betula pumila, Thuja occidentalis, Abies balsamea, Aster borealis, Muhlenbergia glomerata,* and *Onoclea sensibilis.*

Effects on Density, Biomass, Growth, Germination

Biomass of the submersed aquatic plant, *Vallisneria americana,* when transplanted to acidic lakes quickly fell to 0.008 g dry mass, whereas plants transplanted to alkaline lakes grew vigorously to 7.5 g dry weight (Overath *et al.* 1991). In laboratory experiments, tuber growth was decreased by 97% for *Vallisneria americana* at pH 5 compared to normal growth at pH 7.5. At the same low pH, *Najas flexilis* produced no flowers and few tubers (Titus and Hoover 1993). In another round of experiments, low pH (pH 5) reduced the growth of *Vallisneria americana*, but 0.15 mM $KHCO_3$ stimulated growth by 2.8 to 10 fold. CO_2 availbility appeared to be an important control on the growth of this species (Titus *et al.* 1990). Growth of *Carex exilis* seedlings was mostly unaffected by varied acidity (Santelmann 1991). In a laboratory experiment, the germination of *Typha latifolia* seeds exposed to cattail ash, leaf extracts, and a variety of pH levels was unaffected by any of the treatments (Rivard and Woodard 1989).

4.5 Effects of Salinization

Processes

High concentrations of soluble salts in soil water are lethal to plants, and sub-lethal levels may impair growth (Rendig and Taylor 1989). Woody plants tend to be less tolerant than herbaceous plants because they do not have mechanisms for removing salt, other than accumulating salts in leaves and subsequently dropping them.

Effects on Species Composition

Many plant species that inhabit inland saline wetland and coastal tidal wetlands are, of course, adapted to tolerating salt levels that would kill most wetland plant species. A survey of inland lakes in western Canada which spanned a salinity gradient identified relative tolerance to salinity, and specific salinity tolerance thresholds, of many wetland species (Hammer and Heseltine 1988). Short-term salinity pulses (or fresh water pulses) that occur during storm events can affect spatial patterns and species composition of plant communities in tidal marshes (Howard and Mendelssohn 1999b).

Effects on Growth, Germination

The susceptibility of plant species is highly variable with some suffering signs of stress almost immediately whereas others can tolerate elevated salinity for 6-8 weeks (Howard and Mendelssohn 1999a). Salinity concentrations of as little as 3 ppt result in a substantial stress in several southeastern wetland trees (e.g., *Nyssa aquatica*, *Taxodium distichum*) (see review in Pezeshki *et al.* 1990). Biomass of *Taxodium distichum* var. *distichum* seedlings declined after flooding with water of 6-8 ppt salinity (Allen *et al.* 1997) but seedlings survived 10 ppt salinity for 8 weeks in another experiment (Conner et al. 1997). *Cephalanthus occidentalis* and *Nyssa sylvatica* var. *biflora* are both sensitive to salinity in excess of 2 ppt. Responses of these species to salinity are most evident in gross photosynthesis, stomatal conductance, water pressure potential, and stem and root biomass (McCarron et al. 1998).

Among emergent plants, the growth of *Typha domingensis* was reduced to zero when salinity exceeded 6 ppt, and 15 ppt salinity caused 75% mortality (Glenn *et al.* 1995). In contrast, *Carex exilis* seedlings were virtually unaffected by minor differences in salinity (Santelmann 1991).

Some species of plants are stimulated by low salinity levels. *Chenopodium rubrum* growth was stimulated by low concentrations of sodium sulfate and sodium chloride although increased concentrations reduced dry mass and leaf area (Warne *et al.* 1989). *Panicum hemitomon* and *Leersia oryzoides* tolerated 9.4% salinity for up to a month in laboratory conditions whereas *Sagittaria lancifolia* showed damage at only 4.8% salinity (McKee and Mendelssohn 1989).

Effects of salinity may interact with inundation to influence plant mortality. At low salinities, *Leersia oryzoides* growth appears to be inhibited by flooding, whereas *Panicum hemitomon* growth was not significantly affected by flooding (McKee and Mendelssohn 1989). Growth may also be reduced due to the combination of salinity with reductions in soil oxygen associated with flooding, as with seawater flooding of *Taxodium distichum* and *Nyssa aquatica* (Pezeshki 1990, Pezeshki et al. 1995). In southern forested wetlands, *Taxodium distichum* var. *distichum* tolerated flooding with low salinity water (salinity 2 g/L) whereas biomass decreased after flooding with water of 6-8 g/L salinity (Allen *et al.* 1994). *Taxodium distichum* collected from coastal zones of the southern United States were more tolerant to salinity stress than those collected from inland zones (Allen *et al.* 1997).

Salinity influxes may harm the ability of seeds to germinate, even after the salinity stress is relieved (Khan and Ungar 1999). When seeds of *Triglochin maritima* from an inland salt marsh were exposed to various salinity levels, a level of 400 mol/m^3 NaCl resulted in no germination. Even seeds that were then transferred to distilled water suffered some degree of mortality.

4.6 Effects of Sedimentation, Burial

Processes

Sedimentation is a naturally occurring process in wetland systems, but accelerated rates of sediment deposition (or erosion) can tax the ability of wetland plant communities to adapt (Kantrud *et al.* 1989, Jurik *et al.* 1994, Wang *et al.* 1994). Sedimentation can affect wetland systems by the addition of sediment-born pollutants, the burial of established vegetation, and the burial of seed banks (Neely and Baker 1989, Childers and Gosselink 1990, Pucket *et al.* 1993). Burial of leaves has the direct effect of removing light needed for photosynthesis, and restricting foliar gas exchange (Ewing 1996). Buried plants expend energy elongating their shoots in an attempt to outpace sedimentation, seeking oxygen and light, and consequently may be less robust (lower biomass). Over the long term, sedimentation can shrink the proportion of shallow wetlands that remains suitable for wetland plants, or increase the suitable habitat area in ponds that previously were too deep to support many wetland plants. Such long-term changes in water depth (relative elevation) due to sedimentation also result in shifts in species composition, as has been documented in the Mississippi River floodplain.

Moderate rates of sedimentation are also required by several species native to floodplain wetlands. For example, the endangered *Boltonia decurrens*, a perennial member of the aster family, appears to require bare and sandy alluvial deposits for germination and survival in Illinois (Smith et al. 1993, Stoeker *et al.* 1995). In the riparian zones of Alberta, cottonwood seedlings display a high degree of recruitment following a record flood in early June of 1995 (Rood *et al.* 1998). The flooding and the accompanying sediments that were deposited with seeds induced germination of *Populus angustifolia*, *Populus balsamifera,* and *Populus deltoides*. The presence of naturally-deposited islands and bars in large rivers was considered by Dykaar and Wigington (2000) to be a useful indicator of river-floodplain integrity because of the role of these features in sustaining stands of cottonwoods and several other riparian trees.

Effects of sedimentation on particular wetland plant species are not well documented (van der Valk and Jolly 1992, Bartel and Maristany 1989). Many mature plants, and especially woody species, apparently are not harmed by a small amount of sedimentation (Wang *et al.* 1994). Adult plants of *Vallisneria americana* tolerated burial to depths of up to 10 cm but none survived burial under sediment depths of 25 cm (Rybicki and Carter 1986). Growth of the invasive reed, *Phragmites australis*, typically keeps pace with moderate rates of sedimentation (Pyke & Havens 1999). However, seeds, seedlings, and plants that have evolved in wetland types in which sedimentation is rare (*e.g.*, bogs) are highly sensitive to burial. The size of particles that are being deposited, not just their amount, also may influence plant survival (Dittmar and Neely 1999).

Effects on Species Richness, Species Composition

Significant declines in seedling species richness were observed in wetland plots receiving as little as 0.25 cm of sediments (Jurik *et al.* 1994). Deposition (in floodplains) of sediments to a depth approaching 1 m can prevent shallow rooted species from becoming established. Such deposition also can result in a shift to species capable of being sustained only by local precipitation, as well as species unaccustomed to the severe natural disturbance regimes of floodplains (Shafroth et al. 1995).

Sedimentation can result in significant community change as the germination and growth of the most sensitive species are suppressed. Species with larger seeds appear to be better able to survive burial with excessive amounts of sediment (Dittmar and Neely 1999, Jurik *et al.* 1994, Wang *et al.* 1994). Accelerated sedimentation of backwater wetlands was at least partly responsible for allowing a non-native plant, *Colocasia esculenta,* to proliferate (White 1993). Of 14 taxa experimentally buried with sediments of various sizes by Dittmar and Neely (1999), only *Bidens coronata, Polygonum amphibium, Ludwigia palustris,* and a *Carex* species were negatively affected. The seemingly unaffected (or positively affected) species included *Epilobium coloratum, Eupatorium perfoliatum, Galium tinctorium, Impatiens capensis, Lycopus uniflorus, Polygonum pennsylvanicum, Polygonum persicaria, Polygonum punctatum, Polygonum sagittatum and Verbena hastata*. In a survey of several Pennsylvania wetlands where sedimentation rates were also measured, only 6 of the 35 species were considered intolerant of sedimentation (Wardrop and Brooks 1998) (Table 4.2). Although the sedges *Carex rostrata* and *Carex stipata* were mostly resilient to cycles of flooding and drying, sediment deposits resulted in decreased biomass, which was diminished further by high water levels (Ewing 1996). Among woody plants, saplings of *Alnus rubra* tolerated burial less well than those of *Fraxinus latifolia* (Ewing 1996). In Florida wetlands receiving sediment-laden stormwater, several invasive species (*Typha latifolia*, *Ludwigia peruviana* and *Mikania scandens*) were most dominant nearest the input pipe (Carr 1994).

Table 4.2. Sediment tolerance categories assigned to Pennsylvania wetland plants experiencing sedimentation rates of 0 to 8 cm/year (Wardrop and Brooks 1998).

Intolerant
Asclepias syriaca
Aster vimineus
Cirsium arvense
Lysimachia nummularia
Mentha arvensis
Poa pratensis

Moderately Tolerant
Brachyelytrum erectum
Carex emoryi
Carex folliculata
Carex retroflexa
Carex prasina
Carex stricta
Carex vulpinoidea
Phalaris arundinacea
Solidago sp.
Symplocarpus foetidus
Thelyoteris noveboracensis
Triadenum virginicum

Slightly Tolerant
Juncus canadensis
Euthamia graminfolia
Sagittaria latifolia
Eleocharis sp.
Verbena hastata
Equisetum arvense
Carex intumescens
Solidago Canadensis
Utrica dioca

Very Tolerant
Aster novae-angliae
Dipsacus sylvestris
Dulichium arundinaceum
Impatiens capensis
Leersia oryzoides
Polygonum sagittatum
Solidago patula
Solidago uliginosa

Effects on Density, Biomass, Germination

Excessive sedimentation can reduce seedling recruitment (Jurik *et al.* 1994). For example, *Typha* seedling density and biomass decreased as sediment loads increased from 0.2 to 1.0 cm. One study found a fourfold greater density of annuals (vs. perennials) in some heavily sedimented sites (Neely and Wiler 1993). Older and larger seedlings tolerated burial better (Wang *et al.* 1994).

Sediment additions have been found to reduce germination rates of wetland herb species by 34% (Neely and Wiler 1993), 80% (Jurik *et al.* 1994), and 90% (Wang *et al.* 1994). Sedimentation can significantly alter the species composition of wetland plant communities, as seeds of the most sensitive species fail to germinate (Dittmar and Neely 1999). Less than 1 cm of sediment can inhibit germination of *Typha* sp., *Echinocola crusgalli*, *Leersia oryzoides*, and *Carex* sp. (Jurik *et al.* 1994). In contrast, burial by 2 cm of sediment does not interfere with germination of several non-native plant species (Blackshaw 1992, Reddy and Singh 1992). Sedimentation inhibits the germination of *Typha latifolia* seeds more than *Sparganium eurycarum* seeds (Neely *et al.* 1993). *Typha* germination (as well as species abundance and total number of individuals) was inhibited by burial under as little as 0.25 cm layer of sediment; this species germinates best under low oxygen conditions but with light present (Jurik *et al.* 1994). Germination of cattail (*Typha* x *glauca*) seeds decreased by 60-90% when sediment loads of 0.2 to 0.4 cm were applied to the surface of the soil (Wang *et al.* 1994).

Germination of emergent herbaceous species is typically promoted by cold stratification, seeds positioned in the light at the surface of wet but not flooded soils, and fluctuating temperatures (Shipley and Parent 1991). Deposits of sediments on seeds lying on exposed soils can alter these conditions and reduce seed germination. Seeds submersed underwater are also vulnerable to sedimentation, but might be slightly more tolerant (Neely *et al.* 1993, Clevering 1998). Germination of a submersed plant, *Myriophyllum spicatum,* was reduced by sediment depths of 2 cm or more (Hartleb *et al.* 1993).

4.7 Effects of Thermal Alteration

Temperature affects wetland plants mainly by influencing seed germination, and by extending or shortening the growing season. Growth (biomass accumulation) of some species also may be affected (e.g., *Phalaris arundinacea*, Bernard & Lauve1995), giving particular species a competitive advantage or disadvantage (Landhausser and Lieffers 1994). Gradually rising springtime temperature triggers the germination of many wetland herbs (Leck 1989, Hogenbirk and Wein 1992). Abnormally increased temperatures during the dormant season can potentially hinder germination of some seeds that require cold temperatures to alter hormones related with germination. Heated effluents from power plants (Crowder and Painter 1991, Taylor and Helwig 1995), wastewater treatment facilities, landfills (Bernard & Lauve 1995), and other sources can support localized populations of species that otherwise occur farther south (e.g., but may interfere with germination of species . Global climate warming also is anticipated to cause geographic shifts in wetland plant communities.

Germination of wetland plants is affected not only by temperature magnitude, but also by temperature fluctuation (amplitude), at least during particularly crucial times of the year. For example, in experiments involving 45 combinations of diurnal mean temperature and amplitude, Ekstam and Foresby (1999) found *Phragmites australis* to require a high amplitude (>10 °C) for germination over the entire range of mean temperatures, whereas germination of this species was less sensitive to mean temperature than was germination of *Typha latifolia.*

Temperature can also influence species composition by influencing the amount of oxygen persisting in soils or sediments (Callaway & King 1996, Crawford & Braendle 1996). Higher levels of oxygen are typically associated with cooler temperatures, and create an environment that may be more favorable to facultative and upland species than to wetland obligates.

4.8 Effects of Vegetation Removal

Processes

By definition, removal of any vegetation from a plant community causes at least a short-term change in plant biomass and possibly species composition. Removal can occur as a result of fire, tillage, mowing, herbivory (including grazing by ungulates and aquatic animals -- see reviews by Newman 1991, Naiman & Rodgers 1997), mortality from contaminants (e.g., herbicides), logging or beaver activity, dredging or construction activities, or damage from wind (Loope et al. 1994), ice, or flooding.

Vegetation in wetlands typically is adapted for the usual kinds and intensities of disturbances that have occurred for centuries prior to the advent of human influence. That is, many wetlands affected by natural disturbances eventually recover to a condition somewhat resembling their prior state, provided surrounding landscapes have not been drastically changed by humans. For example, fires are historically a natural phenomenon in Carolina bays and pocosins, and no significant change was found in species richness, evenness, or diversity of plant communities following one fire (Kirkman and Sharitz 1994). In other situations, particularly when wetland plants are already exposed to drought, floods, or other severe stress, fire can induce large shifts in species composition and sometimes biomass (Tilman and El Haddi 1992). For example, although fire alone had little effect on the emergence of a fire-tolerant grass (*P. hemitomon),* winter fire followed by spring inundation significantly decreased emergence (Kirkman and Sharitz 1993). Vegetation removal can also facilitate the introduction or expansion of weedy species (see Species Composition, below). Fire can mobilize some contaminants from soils (e.g., boron, Busch and Smith 1993). Removal of vegetation -- especially woody vegetation -- by fire, cutting, or severe floods can result in a rise in local water tables, with consequent implications for plant biomass and species composition (Hodgkinson 1992).

The type of removal process appears to influence the type, duration, magnitude of the effect on plants. When removal is total or nearly total (such as with herbicides), recovery occurs mainly via seedling recruitment. When removal is by non-lethal modes (such as herbivory), recovery often is by vegetative growth. These differential effects of lethal and nonlethal disturbances are partly responsible for vegetation patterns in some wetlands (Baldwin and Mendelssohn 1998).

Effects on Species Composition and Richness

Fire is perhaps the most-studied vegetation removal factor affecting species composition in North American wetlands. Fire suppression can lead to the establishment of longer-lived shrubs at the expense of herbaceous vegetation (Shedlock *et al.* 1993). In Atlantic white cedar wetlands in coastal Massachusetts, the dominant *Chamaecyparis thyoides* became much more common after years of fire suppression, but in the last century light timber thinnings have favored its partial replacement by red maple (*Acer rubrum*). Recently, neither *Chamaecyparis thyoides* nor *Acer rubrum* have regenerated well where an undisturbed cedar overstory persists (Motzkin *et al.* 1993). Cattail stands in the Everglades often expand following fires (Urban *et al.* 1993). The time interval between fires also can shape tallgrass prairie marsh communities (Collins and Gibson 1990, Johnson and Knapp 1995). Annual fires tend to reduce diversity to a greater degree than less frequent fires, due their causing a reduction in forb diversity (Johnson and Knapp 1995).

The effects of fire on species composition and biomass depend significantly on the season when burning occurred, as was found in the Delta Marsh wetlands of Manitoba (Thompson & Shay 1989) and in southeast Missouri wetlands. There, burning of wetlands in spring improved habitat and food conditions for waterfowl, whereas burning in summer benefitted shorebirds and helped keep invasive plant species in check (Laubhan 1995).

In some Louisiana wetlands, herbivory and fire individually affect the structure and composition plant communities, but their effects are not necessarily interactive. No significant differences were found in species richness between herbivory treatments or between fire treatments (Taylor *et al.* 1994). Another Louisiana study found *Spartina patens* to be less resilient to herbivory than *Scirpus americanus* (Broome *et al.* 1995).

The highly invasive, often non-native species that typically colonize sites where vegetation has been removed non-selectively can have profound effects on species composition of wetland communities (Swetnam 1990, Busch and Smith 1993, Crins 1989). Native sedges and grasses responded positively following removal of purple loosestrife from wetlands using herbicides; this was attributed partly to increased sun exposure (Gabor *et al.* 1995). In the Florida Everglades, light transmittance by *Typha* spp. was measured as 15% compared with 65% by *Cladium jamaicense*. Consequently, reduced light available for periphytic photosynthesis was predicted to influence the replacement of sawgrass by cattail (Grimshaw et al. 1997).

After 25 years of regeneration following mining of peat and associated vegetation, four mined areas of an Ontario bog had attained 50% similarity with the species composition of a relatively unaltered bog area (Jonsson-Ninniss & Middleton 1991).

Effects on Density, Biomass, Growth

Burning of a Louisiana lacustrine marsh reduced the aboveground biomass of *Spartina patens* and *Bacopa monnieri*. After recovery, none of the species present before the burn increased in biomass as a result of the burn. Biomass was over 1.5 times greater in the plots that remained unburned than in those that were burned. Biomass was almost twice as great in plots protected from herbivory than in plots subject to natural herbivory after the burn (Taylor et al. 1994). Fires can increase cone production and seedling survival of baldcypress (*Taxodium distichum*) by removing competition (Conner & Toliver 1990, Cook and Ewel 1992, Conner 1993). Several other studies have examined the regeneration of southern bottomland and cypress swamps after thinning (Ewel and Davis 1992), clearcutting (Kennedy and Meadows 1993), and hurricane damage (Putz and Sharitz 1991).

Effects of grazing depend partly on density of grazers, duration of presence in the grazed area, availability of food and water in nearby alternative habitats, and season (Popolizio *et al.* 1994, Clary 1995, Fitch & Adams 1998). Excessive herbivory from deer populations may have caused *Chaemacyperis thyoides* to be replaced by *Acer rubrum* in swamps of the New Jersey Pine Barrens (Stoltzfus 1990). Herbivory by nutria (*Myocastor coypus*) can limit regeneration of baldcypress stands (Brantley and Platt 1992, Myers *et al.* 1995). Short-term grazing of riparian vegetation after more than 30 years of cattle exclusion stimulated growth of herbaceous vegetation (Popolizio *et al.* 1994).

Nonetheless, biomass and production of the herbaceous community as a whole can increase following temporary introduction of grazers at low densities in some riparian and wetland communities (Heitschmidt 1990, Matches 1992, Clary 1995). Also, response of riparian communities to partial vegetation removal depends on the type of wetland or riparian community involved (Clary 1995). For instance, a riparian site dominated by the grass, *Agrostis stolonifera*, which was subjected to mowing (when 10, 5, and 1 cm high) in spring, fall, or both seasons, increased or maintained aboveground biomass as measured the following year. Also, plant biomass in mowed sites dominated by *Carex* stayed the same or decreased, following spring, mid-summer, and late summer cuttings. In one of the three *Carex* dominated sites, forbs increased in response to cuttings at 1 cm, probably due to reduced competition (Clary 1995).

On lakeshores and river margins, ice commonly damages or partly removes wetland vegetation, especially during low water years or intentional winter drawdown (Crowder and Painter 1991, Begin and Payette 1991, Belanger and Bedard 1994, Scott et al. 1997). Emergent species seldom tolerate removal of their tops to the extent that their tops no longer protrude above the water surface, whereas some floating-leaved species do (Middleton 1990).

When wetland plants are removed by dredging or excavation, full recovery of plant biomass in the disturbed areas may require more than 10 months, with vegetatively-reproducing species sometimes becoming more dominant than annuals in the disturbed areas (McKnight 1992). Similarly, perennial species gained dominance over annuals, mosses, and ferns in Montana wetlands that were cultivated (Borth 1998). Also, in a subset of these wetlands that were grazed, species found prior to grazing only in shallow areas and along the upland edge tended to be found more often in deeper waters once grazing was initiated.

4.9 Effects of Turbidity/Shade

Turbidity (i.e., decreased water clarity) almost by definition means decreased availability of light to submersed aquatic vegetation, killing many species. Conversely, increased water clarity may result in increased cover of submersed aquatics (Scheffer *et al.* 1993), which in turn can improve water clarity even further by reducing resuspension of sediments (James and Barko 1990). Nutrient additions often increase phytoplankton growth and consequently turbidity, whereas control of nutrient sources may favor vascular plants and increase light penetration of the water column (Dushenko 1990, Hanson and Butler 1990). Turbidity in wetlands also can result from resuspension of bottom sediments or erosion within a wetland, sediment-laden runoff or channel water sources, or windborne inputs. Many introduced bottom-feeding fish, e.g., carp, stir up sediments and consequently have caused changes in submersed aquatic vegetation cover and species composition (Richardson *et al.* 1995). Severe turbidity typically shifts plant community structure towards floating and emergent species and away from submersed species. Differences in turbidity tolerance exist among submersed species, e.g., *Potamogeton pectinatus* appears to be relatively tolerant of murky waters (Kantrud 1990, Nichols and Lathrop 1994). Propagules of *Egeria densa* growing at 1.8 m depth grew well at suspended solid levels up to 25 g/m^3 in spring and autumn, and 35 g/m^3 in summer (Tanner *et al.* 1993).

Canopy cover can have much the same effect as turbidity, reducing the area of underwater vegetation and some terrestrial plants as well (Pukkala *et al.* 1991, 1993, Sims and Pearcy 1993, Small et al. 1996).

4.10 Effects of Dehydration or Inundation

Processes

Topographic variation on the order of a few centimeters can shape the composition and richness of the plant community by influencing the duration (Dicke & Toliver 1990, Merendino & Smith 1991, David 1996, Vivian-Smith 1997, Silverton et al. 1999), timing (Merendino *et al.* 1990, Squires and van der Valk 1992, Scott et al. 1996, 1997, Gladwin and Roelle 1998), and frequency of saturation (van der Valk 1994, Pezeshki *et al.* 1996, 1997, Smith 1996, Pollock *et al.* 1998) in the root zones of wetland plants. The amplitude and rate of water level **fluctuation** also influences plant species composition, biomass, and germination (Hull *et al.* 1989, Hudon 1997, Shay *et al.* 1999) . This is true even when the substrate beneath the plants is not dewatered. Effects of fluctuations on particular species are influenced partly by oxygen status of the sediments, with anaerobic sediments benefiting the early growth of some species (Spencer and Ksander 1997). Minnesota lakes whose water levels were not artificially manipulated supported much more diverse communities of submersed plants (Wilcox & Meeker 1991). Among 26 Seattle-area wetlands, the degree of seasonal water level fluctuation had no statistically significant effect on species richness in the forested wetlands, but was negatively associated with richness found in emergent and shrub wetlands. Fluctuation during the early spring seemed to have an especially detrimental effect on plant richness in the emergent and shrub wetlands (Cooke & Azous 2000). A lack of water level fluctuation can be just as damaging as excessive fluctuation to some wetland species (Rood and Mahoney 1990). This is because many species need a period of desiccation in order to germinate.

In riverine wetlands, inundation is often accompanied by severe scouring of the substrate by major floods. Such scouring reduces the biomass of many species (at least temporarily) but also allows increased germination of understory or underwater species by reducing dense stands of plants (especially herbs) that otherwise would crowd or shade out seedlings (Streng 1989, Friedman et al. 1995, Spink & Rogers 1996, Osterkamp 1998). The influence of periodic floods on floodplain vegetation is so profound that structural patterns of floodplain vegetation sometimes can be used to indicate the nature of prior flood events (e.g., Hupp 1992). In a river in western Colorado, the duration of flooding was found to be more important than the magnitude of annual peak flow in driving lateral channel migration, which in turn was a major influence on patterns of floodplain vegetation (Richter & Richter 2000).

Wetland plants have many adaptations for coping with prolonged flooding or drought (Rubio et al. 1995). Nonetheless, inundation and/or saturated soil conditions potentially kill plants when sediment oxygen deficits alter plant metabolic processes or allow buildup of substances toxic to plants. Thus, soil texture, which influences soil oxygen levels, can influence the sensitivity of some species to inundation (Wallace et al. 1996).

Inundation also may increase or decrease the exposure of plants to competitors and herbivores (Wilson & Keddy 1991) and cause a shift in the location of plant communities within a wetland (van der Valk et al. 1994). The opposite extreme -- dehydration -- kills plants partly by removing the pathway for taking up nutrients and maintaining tissues, and may also increase or decrease competition and plant exposure to herbivory. Interruption of water corridors between wetlands (e.g., by dams or water diversions) can hinder the spread of seeds of some wetland plants, e.g., *Carex* sp. (Budelsky and Galatowitsch 1999), whereas periodic floods can assist recolonization of denuded areas (Spink & Rogers 1996).

Differences exist among plant species with regard their ability to resist drought and flooding. These differences are related to plant life history and physical characteristics (Earnst 1990, Koncalova 1990, Voesenek *et al.* 1993, Kirkman and Sharitz 1993, Teutsch and Sulc 1997). Specifically, the seed dispersal and germination characteristics of plants may have the greatest effect on the relative abundance of species, according to a model simulation exercise conducted by Ellison & Bedford (1995) using 6 years of data from a southern Wisconsin sedge meadow. The size of seeds and differences in the timing and methods of seed dispersal can help explain the occurrence of some species (Grillas *et al.* 1991). In bottomland hardwood forests, smaller seeded species have more seeds dispersed, more germinants, more established seedlings, but not necessarily more surviving seedlings at the end of one year (Jones et al. 1994). Also, some species, such as cat-tail, are able to keep pace with rising water levels because they are able to rapidly elongate their stem tissue to a greater degree than other species (Waters and Shay 1992, Galatowitsch *et al.* 1999) or sprout adventitious roots (Voesenek *et al.* 1989)

Feedback mechanisms exist, inasmuch as plants themselves, via transpiration, can potentially dehydrate wetlands and increase slightly the magnitude of water level fluctuations (Dube *et al.* 1995). Plants can have the opposite effect as well, sheltering surface water and exposed sediments from the evaporative influences of solar radiation and wind, thus maintaining moisture that benefits their growth.

Effects on Species Richness

Inundation and dehydration have variable effects on plant community richness. In a California subalpine wetland, species diversity within *Carex rostrata, Scirpus acutus,* and *Nuphar polysepalum* communities was highest during dry years whereas biomass was lowest then (Rejmankova *et al.* 1999). In Alaskan riparian wetlands, sites with intermediate flooding were the most species-rich, whereas those with no flooding or high or low flood frequency were species-poor (Pollock *et al.* (1998). Species richness tended to peak (at 100-110 species) when mean flooding frequency reached 11-12 events per year. In the riparian zones of six mid-sized streams in Vermont, an average of 90% of the total species present out to 50 m from the high water mark were found within the first 30 m of the high water mark (Spackman & Hughes 1995).

Water table levels often decline when local or regional groundwater resources are depleted, and plant species composition may change as a consequence, e.g., Sonenshein & Hofstetter (1990), Segelquist et al. (1993), Rochow & Rhinesmith (1992), Stromberg et al. (1996). Water table levels also change when river or lake levels are regulated, e.g., Nilsson *et al.* (1991), Hughes (1990), Gregory *et al.* (1991), Naiman and Decamps (1997), Jansson *et al.* (2000). Many plant species inhabit only wetlands that are inundated briefly, and so can provide a major contribution to regional plant richness in regions where many such wetlands have been destroyed by water table declines or converted to other uses (Hoagland & Collins 1997).

In Texas, April drawdown of wetland water levels produced the greatest species richness (McKnight 1992). In a study of 26 Seattle-area wetlands, wetlands whose contributing watersheds became developed during the multiyear study experienced a decline in plant species richness, whereas urban and rural reference wetlands changed little between years (Cooke & Azous 2000).

Seed bank richness decreases with increasing water depth (Wilson *et al.* 1993, Haukos & Smith 1994) and permanency of flooding (Baskin et al. 1996). Old and young beaver ponds in Quebec were found to have similarly rich and abundant seed banks (Le Page & Keddy 1998).

Effects on Species Composition

Because the database on EPA's BAWWG web site (Adamus and Gonyaw 2000) compiles the literature on hydrology-related shifts in plant species composition, we have limited the discussion of autecological sensitivities in this document.

Species that can move vertically with floodwaters (e.g., *Utricularia vulgaris*, which is not rooted to the substrate), or which grow quickly enough to keep their leaves above water, are better able to flourish with increasing water levels (Murkin *et al.* 1991). Woody plants are particularly sensitive to prolonged inundation (especially >80 days) (Niswander & Mitsch 1995, Toner & Keddy 1997, Sharitz & Gresham 1997). Their seedlings consequently are most affected during years when flooding occurs at or shortly after the beginning of the growing season, or when flooding persists for >40% of the growing season (Toner & Keddy 1997). Annual (as opposed to perennial) species tend to increase proportionately in response to drought and some other severe disturbances (Poiani and Johnson 1989), and species richness tends to be lower where lake and river communities are dominated by annuals (Shipley et al. 1991). A relatively high proportion of plant species that are characteristically wetland "facultatives" (as opposed to obligates), including most woody plants, also suggests relatively dry conditions during at least part of the growing season in a wetland. Species with small, light seeds seem particularly adept at colonizing mudflats exposed during drawdowns and after disturbances (Poiani and Johnson 1989, Ellison and Bedford 1995), and tend to emerge early in the season, thus usually increasing their probability of success due to greater light availability (Toner & Keddy 1997). Successive years of annual drawdowns can favor the spread of many non-native plant species within wetlands (van der Valk 1994). Dominance of a wetland by just a few species is sometimes a sign that the wetland has experienced prolonged drought or drawdown (Wilcox 1995).

Restoration of natural hydrologic regimes to regulated rivers can have dramatic effects on plant species composition (Busch & Smith 1995, Poff et al. 1997). Following a large flood on the Hassayampa River in Arizona, several native species that previously had been uncommon increased (Stromberg *et al.* 1997). With careful planning, restoration of historical flows to regulated rivers, and historic water table levels to isolated wetlands, can allow characteristic native species to resume dominance (Stromberg et al. 1991, Toth 1993, Toth et al. 1995, Briggs & Cornelius 1998, Sher et al. 2000). In the Portland, Oregon, metropolitan area, policies that allowed out-of-kind mitigation of wetland losses resulted in plant communities in constructed wetlands that differed significantly from those present in naturally-occurring wetlands (Magee et al. 1999). Engineering specifications for the constructed wetlands favored plant species more typical of deeper, more permanent water.

Effects on Density, Biomass, Production, Germination, Growth

Inundation or dehydration can either increase or decrease the germination, survival, biomass, biomass allocation (roots vs. foliage), growth, and density (basal area or shoots per unit area) of wetland plants. The effect is very species-specific, and depends on the age (life stage) of the plant, concentration of oxygen in sediments, direction of water level change (rising or falling), duration, depth, and season (temperature/light) of flooding (Busch et al. 1998). Many species have only a narrow “window” in which they can germinate, for example, a few-week period when favorable water levels (or temporary lack of competitors) must coincide with favorable temperatures and acceptable water quality (Rood et al. 1998, Roelle & Gladwin 1999).

Woody plant production is typically higher in flowing-water wetlands with natural flood pulses than in wetlands where water is continuously stagnant (Mitsch et al. 1991). Cumulatively, entire communities can be structured by gross factors such as annual maximum and minimum flows (Auble et al. 1994) or very specific hydrologic sequences, e.g., the last date of the first flood of the season and the first date of the second flood (Toner & Keddy 1997), or prolonged flooding followed by drawdown followed immediately by moderate inundation (in the case of baldcypress). Water velocity, channel pattern, and substrate characteristics also are important to some species (Smith and Wellington 1991). For example, the dominance of cottonwoods over saltcedar can be determined partly by the distribution of flow across the floodplain (Szaro 1990, Howe and Knopf 1991, Cuomo 1992).

Responses to specific hydrologic variables of hundreds of individual species that have been studied are presented in the database at EPA’s web site (Adamus & Gonyaw 2000). Evidence from some studies suggests relative tolerance of water level fluctuations is greatest among several non-native or invasive species (Figiel *et al.* 1991, Haworth-Brockman and Murkin 1993, King and Grace 2000). Among tree species, seed weight may be an indicator of drought tolerance, with heavier-seeded species being more tolerant (Streng *et al.* 1989). However, among herbaceous species, seed weight may not correlate with germination characteristics such as lag time, maximum germination rate, and final germination proportion (Shipley & Parent 1991c).

4.11 Effects of Other Human Influences

Several studies have documented changes in wetland vegetation as a result of general watershed development, without diagnosing in all cases the exact stressor(s) responsible. For example, plant species richness in forested Ontario wetlands was found to be negatively correlated with density of roads within 2 km, but the exact mechanism of impact was not defined (Findlay and Houlahan 1997). Plant species richness also was generally less in urban than in riparian forested streamside corridors near Portland, Oregon (O'Neill & Yeakley 2000). Adverse effects of watershed urbanization on wetland plants were detected in Minnesota (Galatowitch et al. 1998, Mensing et al. 1998, Gernes and Helgen 1999) and western Oregon (Magee et al. 1999, Adamus 2001). Alteration of plant communities of Atlantic white cedar wetlands in southern New Jersey corresponded to the occurrence of roads and housing developments, especially when stormwater was directed into these wetlands (Ehrenfeld & Schneider 1990). Over a period of 150 years, the plant communities in a series of Wisconsin lakes appeared to have lost overall biomass and diversity, and characteristically disturbance-tolerant species became more prevalent. Changes were attributed broadly to dredging, filling, introduction of carp and two invasive plants, herbicides, and shoreline raking to remove nuisance macrophytes. Emergent and floating-leaved species suffered the greatest losses (Nichols and Lathrop 1994).

Physical disturbance of wetland soils during the dry season, such as through tillage, compaction, or excavation, can increase the dominance of invasive non-native species (Morin et al. 1989, Sutton 1996, David 1999, Galatowitsch et al. 1999), as well as destroy much of the viable seed bank (Lee 1991). In central Florida, soil disturbance by feral hogs significantly reduced plant cover and biomass in a broadleaf floodplain marsh, but increased plant species diversity and richness (Arrington 1999). Soil tillage often reduces diversity, including both richness and evenness, as documented in a Carolina bay wetland (Kirkman and Sharitz 1994). The tillage treatment disrupted the rhizomes of perennials more than burning and also facilitated germination of annuals in the seed bank and colonization by several invasive species. Persistent seed banks, perenniality coupled with early sexual maturation, and favorable response to disturbance, were influential in maintaining wetland flora after disturbance. Dominant perennials persisted vegetatively, either above or below ground, and were absent from the seed bank.

Invasive plants, especially non-native invaders, significantly alter the species composition of many wetlands, sometimes even forming nearly monotypic stands. Among the most geographically widespread invaders in North America are *Typha*, *Phalaris* sp., *Lythrum salicaria*, *Phragmites* sp., *Myriophyllum spicatum,* and *Hydrilla verticillata.* Their increased dominance has frequently been viewed as a partial consequence of physical disturbance of soils or water levels within a wetland and/or the surrounding landscape, including accelerated sedimentation, eutrophication, and the construction of mitigation wetlands (Confer and Niering 1992, Magee et al. 1999). In Florida, the invasive vine, climbing hempweed (*Mikania scandens*), thrives along the edges of wetlands whose long-term water levels have risen (Moon et al. 1993). Some invasive species, such as *Phalaris arundinacea* and *Leersia oryzoides*, prevent sedge meadow species from recolonizing when they invade constructed or restored wetlands and (Wienhold and van der Valk 1989, van der Valk et al. 1999). Attempts have been made to quantify the ability of particular wetland species to out-compete or displace others, and to identify traits that could be used as a general guide for identifying such species (Keddy et al. 1994, Twolan-Strutt and Keddy 1996, Keddy et al. 1998).

Removal or decline of non-native species typically results in expanded species richness within wetlands, at least temporarily (e.g., Trebitz et al. 1993). However, other than their adverse impacts on native plant diversity, relatively little is known about consequences of many non-native herbaceous species on wetland functions and attributes such as nutrient processing, carbon cycling, water balance, and wildlife habitat support (Anderson 1995). Research on this topic has recently been initiated for some woody species, e.g., non-native saltcedar vs. native cottonwoods (Stromberg 1998).

Many European studies of restored wetlands are summarized by Pfadenhauer and Klotzii (1996). Long-term studies of constructed or restored wetlands anywhere are rare. In one instance, a reforested gravel pit in Ontario was found to differ from natural forested areas even after 107 years, although it had acquired some similar structural characteristics (Larson 1996). In a series of newly restored wetlands in northern New York, mowing and plowing treatments increased wetland plant establishment, but less so than did salvaging and importing soil from other wetlands (ones being altered) (Brown & Bedford 1997). Plowing also significantly increased the establishment of cattail. Following the importing of soil from wetlands being altered, the species richness and percent cover were actually higher in the newly restored wetlands than in natural reference wetlands.

In contrast, after three years of regeneration, the vegetation of a restored prairie pothole was still not similar to that of a natural wetland which had higher plant diversity (Galatowitsch and van der Valk 1995, 1996a,b). The regenerating wetland had significantly fewer sedge meadow species and more submersed aquatic species than the natural wetlands. The restored wetland also had fewer species represented in its seed bank. Recently restored prairie pothole wetlands in Iowa lacked distinct low prairie and wet meadow vegetation zones (Delphey and Dinsmore 1993). The transition from marsh emergents (*e.g.*, cattail, bulrush) to upland vegetation was often abrupt, perhaps reflecting differences in the exposure gradients of the sites (Shipley et al. 1991b).

In phosphate-mined lands of central Florida, a period of at least 7 years was required before restored wetlands mostly resembled natural wetlands (Crisman et al. 1997). Non-native and floating-leaved species dominated in the years immediately following mining. The frequent abundance of submersed and floating-leaved species in “new” wetlands might be attributed to either the transportation of seeds on the feathers and feet of waterfowl, or shading by a canopy of emergent vegetation in more established wetlands (Galatowitsch and van der Valk 1996). Wet-meadow plants do not readily colonize prairie potholes that were cultivated before restoration (Galatowitsch 1993). A broad series of mesocosm experiments involving manipulation of flooding and nutrients indicated that wetlands with both high plant diversity and low biomass will be difficult to create “from scratch” (Weiher et al. 1996).

Natural seed banks, which are important to maintaining plant species diversity (Leck 1989, Leck *et al.* 1989, van der Valk et al. 1992), can be damaged easily by a variety of human alterations (Keddy *et al.* 1989, Wisheu and Keddy 1989, Wisheu and Keddy 1991). Seeds of wetland plants can be categorized as persistent (remaining viable for many years or decades) or transient (viable for less than one year), but little is known regarding which species characteristically have persistent vs. transient seeds, and the degree to which climate, nutrition, contaminants, and other factors might influence this (van der Valk & Rosburg 1997). Wetland plant communities that are most vulnerable to invasion of non-native species following disturbance include those containing species that have low rates of seed viability and seed dispersal (Reinartz and Warne 1991, Galatowitsch and van der Valk 1994). Many taxa that recolonize bare sites have seeds that are naturally adapted for wind dispersal.

Continuous soil disturbance (such as from compaction and road building) can alter species composition and lead to a decline in both the biomass of native species and alteration of the soil conditions that support them (Ehrenfeld and Schneider 1991). Off-road vehicular traffic caused long-lasting shifts in species composition of arctic tundra wetlands (Felix et al. 1992). Use of all-terrain vehicles also impacted Atlantic coastal plain wetlands, reducing the density of propagules and seed in wetland seed banks, and allowing common rushes to displace rare species (Wisheu and Keddy 1991). Excavation and clearing of gas pipeline rights-of-way through forested wetlands in Florida resulted in increased species richness within the wetland clearings and increased percent cover of non-native species, primarily *Micrantemum umbrosum* and *Paspalum notatum* (van Dyke et al. 1993, Shem et al. 1994). Higher densities of roads in the vicinity of forested Ontario wetlands were associated with diminished plant species richness in the wetlands (Streng *et al.* 1989). Effects of roads on wetland flora and fauna are also summarized by Forman and Alexander (1998). In a study of 19 Seattle-area wetlands, wetlands whose contributing watersheds became developed during the multiyear study experienced a decline in plant species richness, whereas reference wetlands in both urban and rural settings without ongoing development changed little between years (Cooke & Azous 2000). Urbanization, with accompanying alterations to the physical and chemical environment of wetlands, has been associated with altered wetland plant communities elsewhere, as well (e.g., Manny & Kenaga 1991).

4.12 Wetland Monitoring

Spatial and Temporal Variability

Choice of appropriate sample sizes depends on measured variation in the target taxa and metrics. Such coefficients of variation for plants are summarized from various studies at: http://www.im.nbs.gov/powcase/powvariation.html

Spatial and temporal variation of plant species composition at the scale of individual wetlands is influenced by connectivity of suitable habitats (Tabacchi et al. 1990), competition (McCreary 1991, Keddy et al. 1994, 1998), water regimes, water chemistry, and other factors described previously in this chapter. Spatial variation in plant diversity was reported to be greater among less-degraded than among more-degraded streams near Chesapeake Bay (Small et al. 1996). Along transects that cross floodplains, plant richness often peaks midway between the base flow channel and seldom-flooded uplands (Gregory et al. 1991), although in some meandering lowland rivers, richness can be greatest immediately adjoining the channel (Stromberg et al. 1997). Temporal variation (interannual change) in plant species richness in 19 Seattle-area wetlands was reported to be statistically insignificant over an 8-year period (Azous & Cooke 1997). Densities of submersed macrophytes, however, often change dramatically from year to year (Blindow 1992).

Along streams feeding Chesapeake Bay, information derived from rapid determinations of plant species richness along a series of only 5-10 survey lines per stream (each line being 5 meters long and usually 1 m wide, paralleling and touching the channel) was sufficient to distinguish streams which other data showed were polluted from those that were not (Small et al. 1996). When species relative abundance was also considered, predictions of water quality were even more accurate. The best-quality streams averaged 40 species (cumulative) per 20 survey lines. The survey lines were 15 m long and followed the stream bank.

In many wetlands and floodplains, plant richness tends to increase initially as community biomass increases, but at some point begins to decline with continued increases in biomass (Shipley et al. 1991, Tilman 1996). However, biomass alone is usually a poor predictor of species richness in wetlands, often being secondary to more direct effects of environmental factors -- particularly light availability -- and evolutionary history (Gough et al. 1994, Grace 1999).

Techniques

Fundamental to the use of plants as indicators of wetland condition are practical procedures for assessing plant assemblages. With good reason, there is no single accepted approach for sampling wetland and riparian plants. That is because sampling design should depend largely on sampling objectives, i.e., for which particular attributes of plant community structure is information needed, and how quantitative does it have to be? Procedures that are used often when employing plants to characterize wetlands are summarized in Table 4.3, and include the following, applied either independently or in combination:

- Unstructured searches
- Systematic transects
- Random plots
- Stratified plot-based surveys

Table 4.3. Types of procedures commonly used to characterize wetland and riparian plant assemblages

Unstructured searches have the advantage of being the quickest and least restrictive option. They generally involve one skilled botanist walking the entirety of a site while keeping a running list of species noticed (sometimes called a "random walk," Planty-Tabacchi et al. 1996). This approach is applicable when the only objective is to assess plant richness and species composition, not percent cover. Disadvantages include the fact that results are strongly influenced by (a) time spent searching per unit wetland area, (b) size and complexity of the site, (c) keenness and taxonomic skills of the searcher, (d) inaccessibility of parts of the site, e.g., deep water. To improve somewhat the comparability of estimates among different sites, searches can be restricted by time (e.g., 10-minutes per acre, or search until no more species found after 3 minutes of searching) and/or by stratifying the search by recognizable habitats within a site.

Systematic transects are commonly used, especially in research studies, to assess wetland and riparian plant communities (e.g., Winward 2000). Transects consist of sample plots or observation ("intercept") points located, usually at even intervals, along generally straight lines. The transect lines also are usually spaced at even distances apart. They may be oriented perpendicular to the long axis of a site, may radiate from the centerpoint of the site, or be oriented in some other manner intended to span moisture gradients. If the number of transects and/or the number of plots or points per transect is sufficient, information on spatial dominance of particular species throughout the site can be obtained. The even spacing of an inadequate number of transects within a site can be a disadvantage because it will be insensitive to (and may totally miss) important environmental gradients that influence a site's plant communities. The transect may unknowingly follow a linear feature of the vegetation, such as a former farm road that has since overgrown, and as a result a row or column of plots may be spatially correlated, compromising the statistical reliability of the data. These issues can be addressed somewhat by sampling large numbers of plots or points along the transects, and/or by using numerous transects per unit of wetland area. Transect approaches are specified for assessing relative dominance of wetland-associated plants in the Corps of Engineers wetland delineation manual. The manual's procedures, when applied to an average 2-acre site, would require 6 plots. At least 100 plots per site would be required using procedures employed in studies of Portland-area wetlands by EPA's National Health and Environmental Effects Research Laboratory (Magee et al. 1999). Researchers studying restored wetlands in West Eugene, Oregon, calculated that 200 intercept points, spaced equally throughout a site, were required to derive estimates of species cover that are confidently within 5-10% of the true values in their wetlands.

Random plots are typically used at sites perceived either as lacking recognizable environmental gradients, or with highly complex gradients. Standard-sized plots are situated according to X-Y coordinates generated by a random numbers table, or other random number generator. No assumptions are made regarding locations of particular gradients that may influence plant distribution. If the number of plots is sufficient, statistically-sound information on spatial dominance of particular species throughout the site can be obtained.

Stratified plot-based procedures also are sometimes used to assess wetland plant communities. This involves using professional judgment, rather than solely systematic or randomized designs, to situate plots or observation points. One option is to place plots in "representative" locations, using judgment to identify locations that seem most typical of the site. This may encounter problems with repeatability. Another option is to stratify a site according to plant *communities* or *associations* (i.e., commonly-correlated assemblages of species), and then sample each association with one or more plots, located randomly or systematically within each association. This requires judgment to recognize and delimit what constitutes plant "associations," inasmuch as no generally-accepted list exists in many regions. To avoid problems with defining associations, one can place plots to include every plant *species* that appears, from an initial site reconnaissance, to constitute more than a prespecified acreage or percent of the site. To conserve sampling effort, and if a primary objective is to assess species richness, one can select the fewest plot locations that will produce the largest species list. However, this requires careful screening of the site and strategic planning to identify – quite subjectively -- the most complementary and species-rich locations. Finally, one can stratify a site by observable physical and chemical features thought to influence plants, such as shade and/or expected duration of inundation (elevation), and then allocate plots randomly or systematically within each "zone." This also requires judgment to define and locate such gradients, and may have low repeatability when used by different researchers studying the same wetland at different times.

It should be noted that the intensity and statistical rigor appropriate for some plant surveys (e.g., research studies, or periodic monitoring of mitigation sites) is not necessarily appropriate or necessary for studies whose aim is solely to develop plant IBI's or to conduct one-time assessments of relative ecological condition of a series of wetlands. If plant surveys must be limited to only brief and/or infrequent visits to a site (as is often the case when implementing a regional assessment program), it is highly unlikely that enough plots can be surveyed to yield statistically-sound estimates of the percent of the site occupied by each species. This will be true regardless of whether plots are located systematically or randomly. On the other hand, a one-day visit, especially during the growing season and using unstructured searches or stratified plots, will usually be sufficient to determine presence of a large percentage of species occupying a site. However, this implies a tradeoff between speed and repeatability (consistency) of results.

Metrics for Assessing Impacts to Wetland Plant Communities

Species richness (and diversity) can increase (Grassle 1989, Kaczor and Harnett 1990, Phillips *et al.* 1994) or decrease (Brown and Brussock 1991, Englund 1991, Wilson and Tilman 1991) along disturbance gradients, whether natural or human-related. Wetland plant richness also varies latitudinally within continents but shows no clear geographic pattern (Crow 1993). Plant species richness in raised bogs is remarkably uniform along a latitudinal gradient from the northeastern US into central Canada, averaging 20 to 26 species (Glaser 1992). The richest bog community occurs on maritime islands off of the northeastern US, with species richness ranging from 50 in the south to 32 in the northeastern Canadian sites. The southern range of some bog species is determined by maximum summer temperatures which accelerate metabolic consumption rates to lethal levels (Crawford 1989). Thus, to be useful as metrics, richness or diversity should be calibrated regionally, and in any case may not be good indicators of human alterations to wetland plant communities if they are used alone. Data on species composition and other metrics is generally needed as well.

A survey of 22 forested wetlands in the St. Jones watershed of Delaware, using plant species composition and richness as indicators, failed to find strong statistical relationships to apparent watershed condition (Emslie & Clancy 1999). In Minnesota, a wetland plant IBI reflected contamination of wetland water with chloride and sediment contamination with copper (Gernes & Helgen 1999). A statistically weaker response of wetland plants to ambient phosphorus was noted. In western Oregon, the percent of total plant species at a site that are native was found to correlate negatively with several indicators of human alteration estimated at both the site-scale and landscape-scale (Adamus 2001).

Species accumulation curves sometimes can be used to assess degradation and to identify the minimum number of plots per site required to determine this, e.g., by testing for significant differences in slopes of curves from presumably degraded vs. undegraded sites. In streams of the Chesapeake watershed, the cumulative total of shoreline plant species tended to increase slowly as more degraded sites were surveyed, whereas the cumulative total of shoreline plant species increased significantly more rapidly, and reached a higher total, as a comparable number of less-degraded sites were surveyed (Small et al. 1996). Graphs of the number of individual plants per species vs. number of species ("Preston curves") demonstrated separation of altered and unaltered streams. In Chesapeake Bay itself, the distribution and biomass of submersed aquatic macrophytes were found to be a reliable indicator of water quality (Dennison et al. 1993).

For years, botanists have attempted to identify species traits that would allow rapid categorization of species according to sensitivity to specific impacts or to habitat alteration generally. Examples include attempts by Grace (1990), Wood & Tanner (1990), McIntyre et al. (1995), and Boutin & Keddy (1993) as elaborated by Keddy (2000). Such categorizations of species into "guilds" or "functional groups" could greatly expedite the development of successful multimetric indices. However, for an enormous number of wetland plant species, essential prerequisite information about life history is unknown, as is also the case regarding characteristics that reliably distinguish sensitivity to human influences from sensitivity to natural phenomena.

Paralleling the increasing use of plants for assessing wetland condition has been the development of improved conceptual and computer models for predicting long-term sustainability of populations and restored wetland plant communities, given particular hydrologic scenarios for woody plants (e.g., Stromberg et al. 1993, Richter & Richter 2000), herbaceous plants (Shipley et al. 1991a), or wetlands generally (Poiani & Johnson 1993, Weiher & Keddy 1995, 1999, Weiher et al. 1998, Hill et al. 1998). These hold promise for characterizing disturbance gradients across landscapes, as well as for improving water management and the design of constructed wetlands, so that wetland plant communities are better protected. Also, increased attention has focused on use of satellite sensors for detecting stands of stressed vegetation, and especially stands of easily recognizable non-native or invasive species. The potential for use of biomarkers (chemical signatures of individual plants) in detecting wetland alteration also is being explored, e.g., Miller et al. (1993).

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Section 5. Invertebrates

5.1 Use as Indicators

This section describes invertebrates (including insects) that are commonly associated with wetlands, and their response to human-associated changes to wetlands. This section includes many species of (for example) midges, aquatic worms, dragonflies, snails, and water beetles. This section summarizes only the scientific findings on this topic that have been published since 1989. For a general discussion of the topic, and for scientific information published before 1990, readers should refer to Adamus and Brandt (1990).

Of particular note is the book recently published by Batzer et al. (1999) and compilations of information on invertebrates in prairie wetlands (Adamus 1996). Sampling protocols, metrics, and indices have been well developed, at least for stream and lake biomonitoring (Plafkin *et al.* 1989, Klemm *et al.* 1990, Barbour *et al.* 1992, Barton & Metcalfe-Smith 1993, Rosenberg & Resh 1993, Kerans & Karr 1994). However, these are not uniformly transferable to wetlands. Nonetheless, a growing body of literature is addressing the sampling of invertebrates in wetlands. Advantages and disadvantages of using macroinvertebrates as indicators of wetland condition were summarized by Adamus and Brandt (1990).

In the past decade, several studies have used invertebrate assemblages specifically to indicate the ecological condition of a large series of wetlands. Results have been published from such studies, for example, in the Seattle area (Ludwa 1994, Ludwa & Richter 2000), Massachusetts (Carlisle et al. 1998), Montana (Apfelbeck 1998), and Minnesota (Gernes & Helgen 1999, Mensing et al. 1998), and the Great Lakes (Burton *et al.* 1999). Most of these studies are detailed at: http://www.epa.gov/owow/wetlands/bawwg/case.html .

In Minnesota, wetland invertebrates were used to represent the condition of landscapes that contained a large riparian wetland component (Galatowitsch et al. 1998, Mensing et al. 1998). Sampling 15 wetlands belonging to each of 8 wetland types, the investigators found positive or negative correlations of several metrics with a site disturbance score and/or various land cover types measured within 500, 1000, and 2500 m of each wetland. Results are reported by wetland type at: http://www.hort.agri.umn.edu/mnwet/ .

Other efforts to develop wetland indices of biotic integrity (IBI's) using invertebrates are underway in Ohio, Michigan, Maine, Florida, and elsewhere. As a partial starting point for such indices, Adamus and Gonyaw (2001) compiled literature and prepared a documented species database for EPA's internet web site, that categorizes many wetland invertebrates as tolerant or intolerant, with regard to overall sensitivity, and/or specific sensitivity to excessive nutrients and hydrologic alteration: http://www.epa.gov/owow/wetlands/bawwg/publicat.html

Florida Department of Environmental Protection has developed a wetland "Bio-
scores macroinvertebrate taxa present in dip sweeps using a weighted index for sampled taxa. Three metrics are currently computed: total taxa richness, total lake index, and total number of Ephemeroptera, Trichoptera, and Odonata taxa. Wetland health or impairment is determined based on deviation from target values. The scoring and selection of metrics is based on similar procedures for Florida streams and lakes and judgment of local biologists. Florida also is developing a biomonitoring program for canal systems using a modified stream condition index protocol.

In drier wetlands (those not inundated for long periods), it is often impossible to use many of the traditional assemblages of aquatic invertebrates as indicators of ecological condition because these assemblages require permanent inundation. As such, their use would erroneously imply that "good condition" corresponds with "long duration flooding." In drier wetlands, surveys of soil fauna (e.g., earthworms, nematodes), including their dormant stages, have the potential to accurately represent ecological condition (Linden et al. 1994). This has been demonstrated in many European studies (e.g., Goede & Bongers 1994, Korthals et al. 1996, van Straalen & Verhoef 1997, van Straalen 1998, Gyedu-Ababio et al. 1998, Ritz & Trudgill 1999, Bongers & Ferris 1999, and Urzelai et al. 2000) but only recently in some North American studies (Lau et al. 1997, Ettema et al. 1998, 1999).

5.2 Effects of Enrichment, Eutrophication, Reduced Dissolved Oxygen

Processes

Excessive nutrients can cause long-term or short-term shifts in invertebrate community richness, abundance (density), and species composition. These changes are typically triggered when excessive nutrients lead to greater growth of aquatic plants, and in particular the increased dominance of certain kinds of algae. Invertebrate species that happen to be specialized to feed on these algae, or which characteristically find shelter and attachment sites in the aquatic plants, are then favored (Murkin *et al.* 1991, Campeau *et al.* 1994, Moore *et al.* 1993). However, excessive nutrients can trigger severe outbreaks in bacterial taxa harmful to invertebrates; bacterial infestations covering more than 25% of the exterior of individual mayfly specimens were especially lethal (Lemly & King 2000). In addition, respiration and decay of extensive algal biomass can reduce dissolved oxygen in the water column and sediments to levels critical to many wetland invertebrates. Taxa with shorter generation times are especially likely to respond to nutrient increases, and thus maintain a competitive advantage, because the blooms of algae associated with enrichment are often short-lived. In Massachusetts (Carlisle et al. 1998) and Minnesota (Gernes & Helgen 1998), indices of wetland biological integrity using invertebrates were correlated negatively with nutrient-laden stormwater inputs to wetlands. Ammonium fertilizers commonly used in agriculture can be lethal to earthworms (Linden et al. 1994).

Long-term shifts can occur when excess nutrients continue to cycle even after enrichment is abated (Graves *et al.* 1998). Temporary shifts can result from episodic or chronic inputs to wetlands that are less retentive of nutrients (Gabor *et al.* 1994). There may be some level of nutrient input that has no measurable affect on community structure as it lies within the environmental tolerance range of all the major organisms of a wetland's food web (Cooper 1993).

Effects on Species Richness

Up to some point, nutrient inputs to wetlands can lead to increased invertebrate richness, as more food sources become available to predatory invertebrates (Moore *et al.* 1993, Rader and Richardson 1992, Campeau *et al.* 1994, Cieminski & Flake 1995, Gernes and Helgen 1999). However, invertebrate richness in a series of highly enriched wastewater wetlands was found to be lower than in a less enriched reference wetland (Nelson et al. 2000). Excess nutrients in lakes have caused local extinction of some macroinvertebrates and an overall decline in species diversity (Mason 1991), partly due to associated diminuation of dissolved oxygen. Similar responses have been noted in enriched streams (Delong and Brusven 1998). However, a study of Florida lakes found no correlation between nutrient levels and zooplankton (copepod) diversity (Blancher 1984).

Effects on Species Composition

Studies conducted in the last decade continue to strengthen the premise that significant enrichment causes the species composition of wetland invertebrate communities to shift, and that in some cases the characteristics of such shifts can be used to diagnose enrichment as the cause of altered wetland condition. Literature describing the sensitivities of individual species of North America wetland macroinvertebrates to nutrient enrichment has recently and exhaustively been compiled for EPA in a public database (Adamus and Gonyaw 2000). Such information is especially useful because characterizations of invertebrate species according to the trophic state of lakes, or as pollution-tolerant or intolerant in streams (e.g., Hilsenhoff 1982a, 1982b, Rosenberg and Resh 1993, Patrick and Palavage 1994), cannot be assumed to always be accurate for wetlands.

Because the above-mentioned EPA database covers the recent literature regarding species composition shifts, the discussion of that topic here will be limited. Exposure to organic enrichment and eutrophication frequently causes an increase in macroinvertebrate grazers (such as Tanypodinae midges), and herbivores, detritivores, predators, and “miners” that burrow into macrophytes. These are groups that typically increase with increasing growth of periphyton and emergent aquatic plants (Campeau *et al.* 1994). In Florida, Graves *et al.* (1998) noted that oligochaetes, ancylid gastropods, and midges were more dominant in enriched wetlands, while in similar unenriched wetlands (e.g., 8 ppb phosphorus and 783 ppb nitrogen) the dominant taxa were caddisflies (Trichoptera), talitrid amphipods, and a different midge. Differences were possibly due to the intolerance of the latter taxa to low oxygen conditions present in the substrate and water of the enriched wetlands. Also in the Florida Everglades, Rader and Richardson (1992, 1994) reported a greater number of coleopteran species (especially in the Hydrophilidae and Dytiscidae families) in enriched and intermediate areas than in unimpacted sites (total mean annual density of macroinvertebrates at enriched and intermediate sites was 6.1 and 3.5 times greater, respectively, than in the unenriched area). Except for decapods, especially *Palaemonetes paludosus*, the density of each order or class was higher within enriched and intermediate areas. Percent composition measured 2.6 times higher and density of dipterans as 16.2 times greater at enriched and intermediate sites than at the unenriched site. Dominant dipterans at enriched sites were *Dasyhelia* spp., *Goelkichironomus holoprasinus*, *Larsia decolorata*, *Polypedilum trigonus*, *Pseudochironomus* spp., and *Tanytarsus* sp. J. The number of taxa (primarily Chironomidae) did not increase, and was very similar for all sites.

Other studies, specifically focusing on midges, have found that at the subfamily level, Chironominae and Tanytarsinae, which contain hemoglobin and thus are more tolerant of reduced dissolved oxygen levels, appear to replace Orthocladinae as eutrophy increases (King and Brazner 1999). A study of four lacustrine/bay wetlands bordering Lake Michigan also found that midge communities shifted across nutrient gradients. Oligotrophic bays with low conductivity (190-230 μS/cm) were dominated by *Cladotanytarsus* sp., *Orthocladius* sp. and *Heterotrissocladius changi*, whereas more eutrophic bays with higher conductivity (390-450 μS/cm) were dominated by *Chironomus* sp., *Tanytarsus* sp. and *Cricotopus* sp. Based on dosing experiments by Murkin *et al.* (1994) and Campeau *et al.* (1994), it can be surmised that a major shift in invertebrate species composition can occur in northern lacustrine marshes at concentrations of 60-200 mg/L phosphorus and 1600-4000 μg/L nitrogen.

Effects on Total Abundance or Biomass

Recent literature continues to substantiate the tendency of total invertebrate density to increase with increased nutrients, as algal production becomes less of a limiting factor in the invertebrate community (Murkin *et al.* 1991, Campeau *et al.* 1994, Moore *et al.* 1993). Invertebrate populations in nutrient-poor systems, especially in the arctic and subarctic, are especially quick to respond (Hershey 1992, Hiltner & Hershey 1992, Hinterleitner-Anderson et al. 1992, Bartsch 1994). A survey of 20 Nova Scotia lakes also found a positive relationship between zooplankton density (1-5 individuals/m^3) and total phosphorus (5-20 mg/m^3) (Kerekes *et al.* 1990). At least in wastewater systems, total density may continue to increase, or at least not decrease, even when dissolved oxygen deficits that are associated with enrichment become severe (Nelson *et al.* 2000). However, in enriched Everglades wetlands, depletion of dissolved oxygen supplies caused by high oxygen demand appears to be the most important mechanism leading to the decrease in macroinvertebrate abundance (Mason 1991). Similarly, nutient-rich feedlot runoff can be lethal to several invertebrates due to its high oxygen demand (McCahon et al. 1991). A study of 3 Canadian marshes found that enrichment initially caused a reduction in total abundance of invertebrates, but as vegetation became more fully decomposed and oxygen levels rose, so did invertebrate abundance (Gabor *et al.* 1994).

Removal of nutrients can have a cascading effect of the trophic structure of a water body. Following the cessation of sewage inputs to an English lake, phosphorus levels fell from an average of 155 μg/L P to 78 μg/L (Moss et al. 1996). This resulted in an almost immediate drop in the total chlorophyll level and a subsequent drop in *Daphnia* sp. numbers.

Effects on Metal Bioaccumulation

Nutrients appear to influence the tendency of aquatic invertebrates to accumulate heavy metals, and the type of metals that are accumulated. For instance, zinc, iron and manganese concentrations were higher in midges from nutrient-rich wetlands, whereas high copper concentrations were found in midges from nutrient-poor wetlands (Bendell-Young *et al.* 1994). This may be due at least partly to the bioavailability of various metals being influenced by sediment oxygen conditions, which in turn are partly the result of decomposition of algal blooms triggered by high nutrient concentrations.

5.3 Effects of Contaminant Toxicity

The following subsections first review the effects of metals and then the effects of organic and synthetic compounds such as pesticides.

Effects of Heavy Metals

Processes

Heavy metals such as mercury, lead, zinc, copper, and cadmium can be directly toxic to wetland invertebrates, or can impact invertebrate communities by altering the species composition and abundance of algae and aquatic plants upon which invertebrates depend for food and shelter. Wetlands store heavy metals to an even greater degree than lotic systems (Gambrell 1994), so exposure of invertebrates to metals may be greater in wetlands. Even when present at low background levels, metals can easily be bioaccumulated in wetland invertebrates (Hare et al. 1991). Growth, larval development, and reproduction of invertebrates can also be harmed by long-term exposure to sub-lethal concentrations of trace metals (Timmermans 1993). Relatively little is known of the sub-lethal effects and the fate and balance (*e.g.*, metabolization and accumulation) of metal pollutants in freshwater wetlands (Johnson *et al.* 1993). Under some conditions common in wetlands (especially wastewater wetlands), high levels of iron are toxic to invertebrates, both directly and through alteration of habitat structure with floc layers (Rovers 1998).

The extent to which heavy metals are toxic to wetland invertebrates depends largely on the acidity of the wetland and the particular form of the metal involved. Acidic conditions can mobilize and increase the toxicity of some metals, such as cadmium (Wright & Welbourn 1994), and decrease the toxicity of others, such as aluminum (Wren and Stephenson 1991). However, in a British stream, acidic conditions, combined with high aluminum concentrations increased the mortality of the amphipod *Gammarus pulex* and the mayflies *Baetis rhodani* and *Ephemerella ignita* (McCahon and Pascoe 1989). Some metals, such as iron and aluminum, can to some degree protect invertebrates from otherwise toxic effects of heavy metals in acid mine drainage (Whipple and Dunson 199*2*).

Effects on Richness

Declines in aquatic invertebrate richness have been documented in watersheds with greater percent urban land cover and presumably larger loadings of heavy metals, compared with less-urbanized (control) watersheds (Shutes *et al.* 1993, Casper 1994, Winter and Duthie 1998, Gernes and Helgen 1999). A Montana stream exposed to elevated levels of heavy metals from mining operations had significantly reduced species richness (Poulton *et al.* 1995). More than 20 years after cadmium and cobalt discharges to a freshwater marsh in New York were curtailed, invertebrate richness remained lower than at a control (less-polluted) site (Klerks and Levinton 1993). Moderate recovery of invertebrates from metal contamination was demonstrated in the Coeur D'Alene River in Idaho. Over 22 years following cessation of contamination by zinc and other metals, the number of taxa grew from 0 to 18, while the proportion of mayflies, stoneflies, and caddisflies relative to proportion of midges rose (Hoiland & Rabe 1992, Hoiland *et al* 1994). Recovery of aquatic systems from toxic pollutants is summarized by Cooper (1993).

Effects on Species Composition

In general, gastropods, crustaceans, and molluscs are more sensitive than insects to metal exposure (Johnson *et al.* 1993). Amphipods, midges, and mayflies, have been used successfully in field and laboratory studies to detect metal-related sediment toxicity (Burton 1992, Adams *et al.* 1992a, Cain *et al.* 1992, Clements *et al.* 1989a). Some studies show herbivores and detritivores being the most sensitive to metal additions (Kiffney and Clements 1994a, Leland *et al.* 1989), whereas others have reported scrapers being the most sensitive group (Clements 1994).

In general, mayflies and some stoneflies of western streams are sensitive to metals, whereas caddisflies and midges are relatively tolerant (Clements 1994, Kiffney and Clements 1994b, Leland *et al.* 1989, Nelson and Roline 1996). Following abatement of zinc pollution in a Colorado stream, the mayfly *Rhithrogenia hageni* and the dipteran *Pericoma* recovered and became the dominant benthic organism. The mayflies *Baetis* and *Epeorus,* as well as Chloroperlid stoneflies, the caddisfly *Artctopsyche grandis,* and midges did not appear to be significantly affected by the pollution abatement (Nelson and Roline 1993). A Montana stream exposed to elevated levels of metals lacked mayflies and stoneflies, and had more acid-tolerant/metal-tolerant taxa such as the midge *Cardiocladius* sp. (Poulton *et al.* 1995). However, midge richness and density (with the exception of *Procladius* sp. and *Chironomus* sp.) decreased along a gradient of increasing trace metals in the Buffalo River, New York (Diggins and

Stewart 1998). Among the midges, Orthocladinae are known to tolerate moderately elevated concentrations of metals in some situations (Clements 1994) whereas Tanytarsini are sensitive to copper (Clements 1989a). Some mayflies are more sensitive early in their development and become less sensitive as they mature (Kiffney and Clements 1994a, Diamond *et al.* 1992). Freshwater amphipods (*Gammarus pulex*) suffered higher mortality and behavior alterations when exposed to high doses of copper (Taylor *et al.* 1994). An index of macroinvertebrate community sensitivity to copper and other heavy metals was proposed by Clements *et al.* (1992).

Effects on Abundance, Growth, Behavior, Deformities

There was no significant difference in the density of midges and oligochaetes between a polluted freshwater marsh in New York and a control (less-polluted) marsh. Laboratory tests on the midges and oligochaetes from the two wetlands suggested that the oligochaetes from the heavily polluted site were genetically more resistant to cadmium toxicity (Klerks and Levinton 1993). Agricultural drainage water containing arsenic, boron, lithium, and molybdenum and entering the Stillwater Wildlife Management Area in Nevada proved acutely toxic to many wetland invertebrates (Hoffman 1992, Hallock and Hallock 1993a,b). Copper and some other heavy metals appear to be more damaging to aquatic communities in the spring and summer rather than in the fall (Leland *et al.* 1989). It is thought that the summer dosing coincides more closely with hatching of many macroinvertebrates and that early developmental periods may be more susceptible.

Deformities of midge mouthparts have been used as an indicator of heavy metal contamination (Bird 1995) and pollution generally (Lenat 1993). Wetland invertebrates exposed to fertilizer factory waste had an increased rate of developmental deformities (Clarke 1993), perhaps as a result of metals incidentally associated with the fertilizer. However, invertebrates in some Canadian wetlands contaminated with oil did not have an unusually high rate of deformities (Bendell-Young et al. 2000). In Ontario wetlands and streams, morphological deformities in the labial plates of midges were associated with agricultural, industrial, and domestic pollutants (Dickman *et al.* 1992b, Dickman and Rygiel 1996). Incidence of deformities at a control site, upstream from the source of pollutants, was 9% compared to 47% downstream from the source. Similarly, in waters contaminated with coal tar the midge *Chironomus anthyracinus* had significantly higher levels of head structure deformities (Dickman *et al.* 1992a). Another study recommends that measures of both the weight and head capsule width of *Chironomus* larvae can be used as endpoints for toxicity tests to differentiate reduced growth from retardation of instar development (Day *et al.* 1994). Observation of behavioral responses of macroinvertebrates has been proposed as one rapid, cost-effective means of assessing sub-lethal exposures to contaminants (Heinis *et al.* 1990)

Bioaccumulation

In general, higher metal concentrations are found in animals from polluted sediments (van Hattum *et al.* 1993). Moreover, increasing the benthic concentrations of heavy metals, such as zinc and copper, generally increases the concentrations within the invertebrates (Miller *et al.* 1992, Kiffney and Clements 1993). In 3 Canadian lakes, concentrations of heavy metals in the crayfish *Cambarus bartoni* were correlated with sediment/water concentrations (Alikhan *et al. 1*990). In roadside ditches of Louisiana, concentrations of aluminum, lead and cadmium in the crayfish *Procambarus clarkii* were significantly higher than in commercial catches unaffected by road influences (Madigosky *et al.* 1991).

Monitoring the accumulations of heavy metals in invertebrate tissues can be a more accurate measure of the effects of metals than measuring metal concentrations in the water column (Kiffney and Clements 1996). In at least some instances, the aquatic insects that feed on periphyton or detritus accumulate higher concentrations of metals than predatory insects (Kiffney and Clements 1993). The aquatic mite *Limnesia maculata* and the caddisfly *Mystacides* accumulate trace metals from both the surrounding water and from the contaminated midge larvae they feed on (Timmermans *et al.* 1992). Bioaccumulation often affects the condition of individual animals and thus, presumably, population health and abundance. For example, the dry weight of individuals of the mollusc *Anodonta grandis* declined as cadmium concentrations in tissues of the species increased (Couillard *et al.* 1993).

Several factors have been found to influence bioaccumulation rates. In northwestern Ontario, methyl mercury concentrations in tissues of invertebrate taxa from the Odonata, Corixidae, Gerridae, Gyrinidae, and Phryganeidae/ Polycentropodidae exhibited increases in concentrations in response to flooding (Hall *et al.* 1998). Conductivity explained much of the variability in mercury concentrations of crayfish from 13 Ontario lakes (Allard and Stokes 1989). Warmer temperatures caused increased accumulation of copper and cadmium in the isopod *Asellus aquaticus*, whereas lead accumulation decreased with warming temperature.

Acidification did not appear to affect copper, cadmium, and lead accumulation (van Hattum *et al.* 1993). However, aluminum can interact with acidity to produce greater toxic effects. In one study, the combination of acidity (pH 4.9) and aluminum caused the highest invertebrate mortality, and the mortality decreased when aluminum was complexed with citric acid (McCahon and Pascoe 1989). Another study suggests that copper has greater toxicity to macroinvertebrates in water that is acidic and soft than in water that is more alkaline and hard (Clements *et al.* 1989b). Cadmium concentrations in freshwater clams were most closely associated with water column pH in 21 Ontario lakes (Campbell and Evans 1991). Cadmium absorption by a freshwater clam, *Unio pictorum*, was influenced by cadmium concentration and water temperature. The process of cadmium accumulation by freshwater clams is rapid and appears to be irreversible (Jenner *et al.* 1991).

Acidic conditions can also affect bioaccumulation of metals. Midges from acidic lakes had markedly higher concentrations of aluminum, cadmium, magnesium, and zinc in their tissues than midges from less-acidic lakes (St. Louis 1993).

In general, wetlands receiving acid-mine drainage with low pH and large amounts of dissolved minerals (aluminum, copper, iron, manganese, and zinc) have considerably lower macroinvertebrate abundance and diversity (Short *et al.* 1990). After reclaiming a wetland at an abandoned mine site, the acidity and toxicity of metals decreased and the abundance of aquatic invertebrates increased (Fucik *et al.* 1990). When acid-mine drainage was allowed to reenter the wetland, the aquatic invertebrates decreased in abundance as water quality deteriorated. Aquatic invertebrate populations appeared to be better indicators of the degree of recovery than water samples.

Macroinvertebrate communities in streams exposed to mine tailings and seepage from a molybdenum mine had somewhat lower diversity and richness than control sites, and tended to be dominated by pollution-tolerant species (Whiting *et al.* 1994). Platyhelminthes (*Polycelis* sp.), Ephemeropterans, Trichopterans, and chloroperlids declined in abundance in sites receiving mine tailings. In contrast, the orthoclad midges *Eukiefferiella* sp. and *Parametriocnemus* sp. exhibited the opposite trend. The sites immediately downstream from the tailings had larger numbers of tubificid worms, psychodids and tipulid larvae. In general, collector-gatherers made up a larger proportion of the community in contaminated sites, while grazers and predators were less prevalent (Whiting *et al.* 1994). Sites receiving open mine pit drainage, appearing to contain insufficient molybdenum and cyanide to severely impact the invertebrate community, had somewhat higher densities and richness than control sites. The mayfly *Ameletus* sp. and the caddisflies *Allomyia* sp. and *Chyrandra* sp. were absent from the sites that received open pit drainage. The open pit drainage sites were dominated by naidids, Hydracarina, Ostracods, Amphipods, hydroptilids, and ceratopogonids, which were all rare or absent at control sites. It is unclear why the open pit sites had higher richness, although it is thought that it might contain higher aqueous nitrogen concentrations that led to increased algal growth.

Effects of Pesticides, Oil, and Other Contaminants

Processes

These substances can alter community structure by (a) being acutely or chronically toxic to invertebrates, (b) altering algal communities and aquatic macrophytes upon which some invertebrates depend for food and shelter, (c) altering predation on invertebrates by decimating numbers of other crustaceans, fish, and amphibians, (d) reducing available dissolved oxygen (i.e., chemical oxygen demand) or oxygen diffusion rates (e.g., oil), and (e) altering the effects of other potential stressors, such as acidity. Macroinvertebrates can bioaccumulate some complex pesticides. For example, *Chironomus riparius* bioaccumulates flouranthene and benzo[a]pyrene (BAP) (Clements *et al.* 1994).

Toxicity Effects

With the advent of the mosquito-borne West Nile virus in parts of the United States, nearly all wetlands in some localities are being dosed with non-selective hydrocarbon pesticides. Few field trials of these pesticides have closely examined biological effects in wetlands, so inference must be made from the limited number of laboratory and stream studies.

In laboratory tests the insecticide **diflubenzuron** was most toxic to crustaceans, followed by mayflies, midges, and caddisflies (Eisler 1991b, 1992). Showing moderate sensitivity were the larvae of corixids, dragonfly adults and larvae, spiders, dytiscids, and ostracods. Also lethal to invertebrates were **paraquat** (Eisler 1990), **cyanide** (Eisler 1991a), **fenvalerate** (Eisler 1992b), and **acrolein** (Eisler 1994). Following **toxaphene** application in an Alberta lake, the zooplanktonic invertebrate *Bosmina* sp. was reduced by 88% (Miskimmin and Schindler 1994). The synthetic pyrethroid insecticide **fenvalerate**, when present in saltmarsh sediments, did not cause significant mortality to any life stage of the copepods *Microarthridion littorale, Paronychocamptus wilsoni,* and *Enhydrosoma propinquum*, even after 7 days of exposure. However, fenvalerate concentrations as low as 25 ppb depressed egg production by 50 to 100% and mean clutch size by 40-100% (Chandler 1990). Thus, although a pesticide may not have an immediate effect on a community, restriction of reproductive capacity may lead to a decline in abundance. **Chlorothalonil**, an agricultural fungicide, did not appear to significantly harm *Limnephilus* sp., *Pisidium* sp., *Haliplus* sp., *Gammarus* sp. and midges. However, *Sigara alternata* did experience increased mortality (Ernst *et al.* 1991). **Phorate**, an organophosphorous insecticide, can result in significant mortality of aquatic macroinvertebrates even when applied at recommended rates. In the Prairie Pothole Region, macroinvertebrates that were particularly sensitive to phorate included hemipterans, mosquitoes, flies, mayflies, water mites, and water beetles. Less sensitive were leeches, snails, aquatic worms, and ostracods (Dieter *et al.* 1996). The insecticide **esfenvalerate**, when applied to 12 small Alabama ponds, significantly reduced populations of invertebrates, although rotifers seemed to be less affected (Webber *et al.* 1992). The lampricide 3-trifluoromethyl-4-nitrophenol (**TFM**) had a significant affect on invertebrates in a small stream (Lieffers 1990). **Fenithrothion,** applied for forest insect control, reduced emergence of aquatic insects for 6-12 weeks. Densities of most invertebrates, and especially predatory species, midges, and some other dipterans, were reduced by as much as 50% for more than one month after treatment. The wetland sediments came to be dominated by aquatic worms and water mites. Although in many streams and large lakes fenithrothion has transitory affects, some residual toxicity remained in bog wetlands during the winter and into the next year (Fairchild and Eidt 1993). Effects on invertebrates of insecticides used in rice fields are reviewed by Roger (1995).

In a stream contaminated by various organic pollutants, many mayflies, stoneflies, and caddisflies decreased in abundance (Hachmoller *et al.* 1991). The heptageniid mayflies and most stoneflies were especially sensitive to organic pollution. A few mayfly genera, such as *Baetis* sp. and *Paraleptophlebia* sp., were more tolerant to pollution or at least were more widespread. Of caddisflies, *Hydropsyche* sp. was more common in the unpolluted section of the stream and *Parapsyche* sp. was more common in polluted sections. Mussels are especially sensitive to the combined effects of pesticides, organic compounds, and excessive nutrients (Keller 1993, Metcalfe & Chaarlton 1990). The chronic release of a **timber preservative** that contained three pesticides drastically reduced invertebrate diversity in a river in England (McNeill 1989). In a Pennsylvania stream subjected to industrial pollution, including contamination with PCB's, isopods, oligochaetes, and craneflies were the main survivors, compared with non-urbanized control segments (Kemp and Spotila 1996). After 25 days, an **oil spill** in a Missouri stream had reduced the macroinvertebrate population to less than 0.1% of normal densities. Recovery of some species of stoneflies, mayflies, and caddisflies did not occur until at least nine months later (Crunkilton and Duchrow 1990). In another study, the burrowing mayfly (*Hexagenia* sp.) had reduced densities where sediments contained visible petroleum oil residues (Schloesser *et al.* 1991b). **Vinyl chloride** discharges from a factory severely degraded the macroinvertebrate population in the Niagra River watershed in Ontario (Dickman and Rygiel 1992). Effects of polychlorinated dibenzo-p-dioxins (**PCDD**'s) and dibenzofurans (**PCDF**'s) are reviewed by Fletcher & McKay (1993). Effects of the ingredients of common **detergents** are reviewed by Lewis (1991), who concluded that detergents using linear alkylbenzene sulfonate (LAS) surfactants are perhaps among the more benign ones, in terms of aquatic biological impacts. In laboratory tests, a surfactant was found to be approximately 100 times more toxic than the herbicide glysophate, with which it is commonly applied (Henry et al. 1994).

A biological control agent -- *Bacillus thuringiensis* var. *israelensis* (**B.t.i.) --** appears generally to have minimal adverse effects on non-target insects in streams (Wipfli and Merritt 1994, Kreutzweiser *et al.* 1994a) although mortality has been observed in Lepidoptera (Jackson *et al.* 1994), some midges (Merritt *et al.* 1989), craneflies (Wipfli and Merritt 1994, Waalwijk *et al.* 1992), caddisflies, and mayflies.

5.4 Acidification

Processes

Acidification can alter community structure by (a) being acutely or chronically damaging to tissues of invertebrates – species that easily lose sodium ions when pH is reduced tend to be most sensitive (Steinberg and Wright 1992), (b) altering algal communities and aquatic macrophytes upon which some invertebrates depend for food and shelter, (c) altering predation on invertebrates by decimating numbers of other crustaceans, fish, and amphibians, (d) altering the bioavailability of some other potential stressors, such as heavy metals (Brett 1989, Stokes *et al.* 1989, Feldman and Connor 1992, Stephensen *et al.* 1994). The effects of acidity also depend on the seasonal life cycles of macroinvertebrates and water temperature (Pilgrim and Burt 1993). In areas with snow, the greatest acid stress often occurs during snowmelt. Young larvae were more susceptible than older larvae at that time (Gorham and Vodopich 1992). Metals and acidity also can interact to alter the toxicity of either or both (e.g., Havens 1994a).

Effects on Richness

Acidity often decreases the richness of macroinvertebrates in aquatic habitats (Schell and Kerekes 1989, Hall 1994a). Reductions in acid emissions from some Canadian smelters was followed by significant increases in richness of invertebrates in water bodies downwind of the smelters (Griffiths and Keller 1992). Invertebrate richness in a survey of 20 streams increased as pH increased from 4.2 to 5.7. Above pH 5.7, a reversal occurred and richness decreased, at least when aluminum levels also decreased and humic content increased (Kullberg 1990). Montana wetlands with naturally low acidities were found to have greater dominance by a few taxa, and greater percent Amphipoda and Hilsenhoff Biotic Index. Wetlands with circumneutral pH had greater taxonomic richness, number of Chironomidae taxa, and percent filterer-collectors (Stribling et al. 1995).

Effects on Species Composition

Acidification effects on aquatic invertebrate communities have been researched extensively. Much of the information from European literature (including autecological responses) is compiled by Johnson *et al.* (1993). Researchers in Wales even developed a dichotomous key based on invertebrate indicator species to classify and rank streams according to their acidity (Wade *et al.* 1990). Derived mainly the North American literature, Table 5.1 categorizes some taxa as more or less tolerant of acidification. It should also be noted some invertebrates are sensitive to pH increases. For example, stormwater input to a Florida freshwater marsh increased phosphorus levels, lowered oxygen levels, and raised pH and hardness, resulting in macroinvertebrate population shifts toward species that otherwise are intolerant of typical acidic and oligotrophic conditions in the studied wetland (Graves et al. 1998).

Table 5.1. Relative Tolerance of Invertebrate Taxa to Acidification

<u>More Tolerant (Less Sensitive):</u>

some Odonata, at least damselflies (Parker et al. 1992, Johnson *et al.* 1993, Baker and Christensen 1989)

some water beetles (Parker et al. 1992, Johnson *et al.* 1993) especially hydrophilid and dytiscid beetles (Baker and Christensen 1989)

some water bugs, at least Notonectidae, Gerridae, Corixidae (Johnson *et al.* 1993, Baker and Christensen 1989)

some caddisflies: *Cheumatopsyche pettiti* (Camargo and Ward 1992).

some Megaloptera (*Sialis*)

some Diptera, at least phantom midges (*Chaoborus*) (Johnson *et al.* 1993), midges (Havens 1994a, Baker and Christensen 1989, Tuchman 1993), blackflies (Baker and Christensen 1989).

some stoneflies (Tuchman 1993) such as *Amphinemura* and *Leuctra* (Griffith *et al.* 1995) the mayfly *Eurylophella funeralis* (Griffith *et al.* 1995)

some water mites (Havens 1994a)

some zooplankters, such as *Daphnia galeata mendotae*, *D. retrocurva*, *Skistodiaptomus oregonensis* (Havens 1993) and the rotifers *Gastropus stylifer, Keratella taurocephala, Polyarthra renata, Symchaeta* sp. (Fore *et al.* 1998)

scrapers and collectors (Smith *et al.* 1990); filter feeders (*Pisidium casertanum*) and predators (*Enallagma* sp.) (Mackie 1989)

Less Tolerant (More Sensitive):

some water bugs (Parker et al. 1992)

some caddisflies (Parker et al. 1992), such as *Lepidostoma* sp. (Hall 1994a), and some in the scraper and predator guilds (Williams 1991)

some midges, such as *Tanytarsus, Microtendipes, and Nilothauma* (Griffiths 1992), *Micropsectra* sp. (Hall 1994a)

some blackflies, such as *Simulium* sp., *Prosimulim* sp. (Hall 1994a)

some mayflies (Balding 1992, Steiner 1993), especially *Baetis*, *Serratella*, *Drunella*, *Epeorus*, *Paraleptophlebia* (Melack and Stoddard 1991, Hall 1994a)

some Odonata (*Enallagma civile*, Giberson and MacKay 1991)

Dugesia dorotocephala (Camargo and Ward 1992)

molluscs (Grapentine and Rosenberg 1992, Gibbons and Mackie 1991, Balding 1992), including clams (Schell and Kerekes 1989, Melack and Stoddard 1991) and mussels

snails, leeches (pH >5.0, Schell and Kerekes 1989)

the amphipod *Hyalella azteca* (Havens 1994a, Mackie (1989; pH must remain above 5.8, Grapentine and Rosenberg 1992)

Gammarus minus (Griffith *et al.* 1995)

many stoneflies, e.g., *Peltoperla arcuata* (Griffith *et al.* 1995)

shredders (Tuchman 1993) and deposit feeders (Smith *et al.* 1990)

Bosmina longirostris (Havens 1993)

the rotifers *Asplanchna priodonta, Collotheca mutabilis, Conochiloides sp., Conochilus unicornis, Gastropus hyptopus, Kellicota longispina, Keratella cochlearis, Keratella crassa, Polyarthra dolichoptera, Trichocera cylindrica* (Fore *et al.* 1998)

Effects on Abundance, Density, Biomass, Productivity

With increased acidity, many aquatic invertebrates declined in numbers and biomass, especially in wetlands with pH < 5.0 (Parker *et al.* 1992). Invertebrate densities in a pH 4.5 channel were significantly lower than in channels of pH 5.9 or 7.4 (Griffiths 1992). Reductions in acid emissions from some Canadian smelters were followed by significant increases in densities of invertebrates in water bodies downwind of the smelters (Griffiths and Keller 1992). In many aquatic habitats the abundance and biomass of macroinvertebrates seems to be controlled more by nutrient availability than by acidity (Schell and Kerekes 1989).

5.5 Effects of Salinization

Salinity and specific conductance (conductivity) can markedly influence species composition, richness, and abundance of invertebrates, particularly along coastal rivers and among non-riverine inland wetlands. Effects of salinity on invertebrate communities are perhaps less noticeable in inland streams and rivers, where the range of salinity is less (Williams *et al.* 1991). Salinization of wetlands can occur as a result of cutting off wetlands from some types of groundwater inflow (Swanson et al. 1988), from increasing evaporative water loss, or from discharge of effluents (especially irrigation return water), or from routing runoff of relatively high conductivity into wetlands. Among some lacustrine wetlands in Wyoming, relative production of different invertebrate functional groups (scrapers, deposit-feeders) varied depending on salinity and associated floating macrophytes (*Chara* at low salinity, *Potomogeton* at higher), but total invertebrate production did not. At lower salinity, scrapers and epiphytic deposit-feeders predominated whereas at higher salinity, filter-feeders and benthic deposit-feeders assumed a larger proportion of the total biomass (Hart & Lovvorn 2000).

Processes

High levels of salinity can alter structure of freshwater invertebrate communities by (a) being acutely or chronically damaging to tissues of invertebrates, (b) altering species composition and structure of algal communities and aquatic macrophytes upon which some invertebrates depend for food and shelter, (c) altering predation on invertebrates by decimating numbers of other crustaceans, fish, and amphibians, (d) altering the bioavailability of some other potential stressors, such as heavy metals and nutrients.

Effects on Species Richness

Even at low concentrations, increases in chloride (a correlate of salinity, and often associated with road salt applications) among 27 Minnesota wetlands were significantly correlated with declines in species richness among the wetlands (Gernes and Helgen 1999). In a survey of East African lakes, zooplankton taxa richness began to decline at a conductivity of 1000 μS/cm and declined to just 2-3 rotifer taxa at salinities above 3000 μS/cm (Green 1993).

Effects on Species Composition

In Wyoming wetlands of fairly low salinity (0.8 – 30 mS/cm), the dominant macroinvertebrates are amphipods and epiphytic snails. Above 30 mS/cm, they are mostly replaced by midges, as well as predatory dragonflies and water bugs (Wollheim and Lovvorn 1995). In Utah, as salinity in the Great Salt Lake declined from 13-23% to 6%, the brine shrimp *Artemia salina* and the brine fly *Ephydra cinerea* declined in abundance while *Ephydra hians* increased in dominance (Stephens 1990). Table 5.2 summarizes some of the recent literature on invertebrate salinity preferences and tolerances, categorizing some taxa as more or less tolerant of salinity levels atypical of the habitats they normally inhabit. Other recent species-specific salinity data for wetland invertebrates are presented by Walker et al. (1995), Parker and Wright (1992), and Lovvorn et al. (1999). In Minnesota, as chloride increased across a spatial gradient of 27 wetlands, dragonflies, mayflies, and caddisflies decreased significantly (Gernes and Helgen 1999).

Table 5.2. Relative Tolerance of Freshwater Invertebrate Taxa to Salinity

More Tolerant (Less Sensitive):

Artemia franciscana (Wurtsbaugh and Berry 1990)

Trichocorixa reticulata, *Tanypus grodhausi* (conductivity >53.5 mS/cm, Euliss et al. 1991)

Trichocorixa reticulata, *Tanypus sp.*, *Ephydra* sp., *Brachionus plicitilis* (salinity of 7,367 to >70,000 mg/L TDS, Parke and Knight 1992)

Dolichopodidae, *Ephydra hians* (salinity >63 ppm, Hammer *et al.* 1990)

Ephydra, *Culicoides* (salinities >10%, Short *et al.* 1991)*Bezzia* sp., *Hygrotus salinarius*, *Cricotopus ornatus* (salinity 11-63 ppm, Hammer *et al.* 1990).

Lymnaea elodes (at conductivity >5,000 µS/cm, Swanson et al. 1988)

Less Tolerant (More Sensitive):

Gammarus sp., *Gyptotendipes* sp., *Chironomus plumosus* (salinity 3-10 ppm, Hammer *et al.* 1990)

Branchionus sp., *Cletocampus* sp., *Diaptomus* sp., *Trichocorixa verticalis* (Wurtsbaugh and Berry 1990)

Lymnaea stagnalis (conductivity <5,000 µS/cm, Swanson et al. 1988)

Toxicity Effects

High salinity in irrigation water entering the Stillwater Wildlife Management Area in southwestern Nevada was toxic to *Hyalella azteca* amphipods and *Daphnia magna* (Ingersoll *et al.* 1992). The mussel and the quagga mussel (*D. bugensis*) can survive over 18 days in salt concentrations higher than 5‰ (Spidle *et al.* 1995).

5.6 Sedimentation/ Burial

The accelerated covering of plants and other natural substrates with inorganic particles (i.e., sedimentation) can result in reduced invertebrate richness and density, and alteration of species composition (Hellawell 1986). .

Processes

Exposure to sediments affects behavior and survival of individual invertebrates (Taylor and Pascoe 1994), and consequently invertebrate communities. Because wetlands are basically depositional environments, many wetland invertebrate communities are able to tolerate occasional deposition of small amounts of sediment, whereas constant or severe deposition causes major changes. Excessive sedimentation affects invertebrates because it (a) buries essential detrital and algal food sources, and excess time required to move through deposited sediment and collect scarce food items from a younger substrate may result in lower survival, (b) reduces flow of interstial water necessary to supplying invertebrates with adequate dissolved oxygen, and (c) kills macrophytes that otherwise provide attachment structures and shelter to invertebrates (Hellawell 1986, Newcombe and MacDonald 1991, Ryan 1991). Sediments often contain metals and other contaminants at toxic levels. Approaches for characterizing toxicological and physical risks of sediments to invertebrates are summarized by Adams *et al.* (1992). Once deposited, sediments can further damage wetland invertebrate communities if they are resuspended by wind mixing or fish, making water turbid. For example, benthic feeding carp, *Cyprinus carpio*, noticeably increase water column turbidity both directly (as they move along the bottom) and by consuming aquatic plants that otherwise would stabilize and trap sediments (Lougheed *et al.* 1998). Planktonic invertebrate biomass declined in Utah ponds after introduction of carp (Huener and Kadlec 1992).

Effects on Richness, Abundance, and Density

Prolonged (>10 years) siltation of backwater lakes in the Mississippi River (Iowa) at rates of 1.5 cm/year led to significant declines in populations of the mussel, *Musculium transversum* (Eckblad and Lehtinen 1991). Deposition of 5-10 cm of sediment in willow wetlands in northeastern Missouri resulted in lower invertebrate community richness and density, compared with non-sedimented sites (Magee 1993). In some instances, invertebrate density and perhaps richness can increase over the long term if sedimentation of coarser-particled substrates creates fine-particled substrates that better support establishment of rooted plants. In temporarily flooded prairie pothole wetlands, only caddisflies seemed relatively unaffected by surrounding land use, whereas ostracods, cladocerans, and snails (planorbiids, lymnaeids, physids) were diminished, presumably in part due to the effects of sedimentation (Euliss and Mushet 1999). Across a gradient of increasing land use intensity in Minnesota, the snail *Physa* decreased (Gernes and Helgen 1999).

Effects on Species Composition

Burrowing, tube-forming worms and midges commonly predominate where sediments accumulate (Magee 1993). Filter-feeding and bottom-grazing taxa are most sensitive (Evans 1996, Lougheed and Chow-Fraser 1998). However, invertebrate size and behavior also influence tolerance (McClelland and Brusven 1980). Taxa that characteristically occupy the water column, and especially the smaller forms of such taxa, tend to be less sensitive to sediment deposition than benthic or epiphytic taxa, but may still be highly sensitive to turbidity (Newcome and MacDonald 1991). Substrates newly created by sedimentation may attract tolerant individuals and species that are poor competitors on older, more crowded substrates (Soster and McCall 1990).

Some studies (e.g., Hogg and Norris 1991, Ludwa 1994, Lamberti and Berg 1995, Carlisle et al. 1998, Ludwa & Richter 2000) have linked changes in invertebrate communities to development of watersheds, and development often is accompanied by increased export of sediment to water bodies. Some water beetles (e.g., *Stenelmis crenata, Optioservus fastiditus*), mayflies (e.g., *Baetis tricaudatus, Stenonema* sp.), and even some stoneflies (e.g., *Taeniopteryx nivalis*) can increase in response to watershed development despite the accompanying sedimentation (Lamberti and Berg 1995). However, severe and rapid sedimentation is inevitably lethal to nearly all aquatic invertebrates. In North Dakota, wetlands surrounded by cropland were virtually devoid of the resting eggs of zooplankton, whereas such eggs were present extensively in wetlands surrounded by mostly natural grassland, which presumably minimized erosion and sedimentation (Euliss and Mushet 1999).

Unionid mussels are one group that is sensitive to increased sedimentation (Goudreau et al. 1993, Box and Mossa 1999). Numbers of the mussel, *Musculium partumeium*, and amphipods were reduced in willow wetlands in northeastern Missouri where 5-10 cm of sediment had been recently deposited (Magee 1993). However, some bivalves seem to tolerate moderate levels of turbidity and periodic sedimentation. For example, in a laboratory experiment where the mussels *Amblema plicata plicata* and *Fusconaia ebena* were exposed to total suspended solids up to 120 mg/L for five minutes every 0.5 and 3.0 hours, no consistent effects were observed (Payne and Miller 1999).

In a tidal freshwater wetland, several taxa were relatively unaffected by single depositions of large amounts of sediment (0.3 meters of fluid mud). These included *Limnodrilus* sp., *Limnodrilus hoffmeisteri*, *Ilyodrilus templetoni*, *Corbicula fluminea* and *Coelotanypus scapularis* – all species that typify this dynamic type of wetland. Somewhat greater depths of deposition increased the mortality of midges and small (< 10 mm) *Corbicula fluminea* (Diaz 1994).

5.7 Vegetation Removal

Vegetation has been shown to be a major factor shaping wetland invertebrate communities (Krieger 1992, Wissinger 1999). Indeed, wetland managers often manipulate vegetation structure -- e.g., by mowing, burning, plowing, planting -- to encourage or discourage populations of desirable or undesirable invertebrates (Batzer and Resh 1992a, Kirkman and Sharitz 1994, de Szalay et al. 1996, de Szalay and Resh 1997, Gray et al. 1999).

Processes

Removal of aquatic or riparian vegetation affects invertebrates because it (a) removes attachment substrates that otherwise provide additional vertical space in the water column for colonization, (b) removes shade, thus increasing water temperature and enhancing growth of algae, (c) increases water circulation and perhaps velocity, with accompanying increases in dissolved oxygen and possible resuspension of sediments, (d) reduces inputs of leaf litter that provide food to some invertebrate taxa, (e) reduces structures that otherwise shelter invertebrates from predators (Jordan et al. 1994) and erosive forces (Roman *et al.* 1994), and (f) reduces a source of dissolved oxygen, i.e., plants that are capable of oxidizing sediments.

Effects on Species Richness

Wetland emergent and submerged plants often support a higher richness of taxa than open water areas and sediments, at least during some seasons (Brady 1992, Brady and Burton 1995). Complete removal of vegetation generally reduces richness of the wetland invertebrate community, but patchy removal or moderate grazing sometimes increases richness (McLaughlin and Harris 1990, Gray et al. 1999), such as where rights-of-way cross forested wetlands (King et al. 2000).

Effects on Species Composition

Logically, the taxa that are most closely associated with vascular plants are the ones likely to be adversely affected by vegetation removal. These primarily include algae-feeding species (grazers) and their predators, although removal of part of a forest canopy can result in increased dominance of algae-feeding species (King et al. 2000). Also, over the long term, removal of vegetation from wetlands with little input of leaf litter from uplands can decimate detritivorous taxa.

In a Washington pond, *Helobdella* leeches, *Asellus* isopods, mayflies, and some dragonflies (especially the large-bodied *Anax*) were more commonly associated with emergent vegetation than with submerged vegetation or open water areas. Midges, freshwater shrimp (*Hyalella azteca*), and molluscs (especially *Lymnaea* sp., *Gyraulus* sp., and *Anodonta* sp.) were more common on the submerged plants (Parsons and Matthews 1995). A decrease in early season flooded plant cover can reduce mosquito larvae (Wood et al. 1992) and confine remaining larvae populations to the perimeter of the marshes (Batzer and Resh (1992a). Such a reduction in plant cover also can increase the density of midges and dytiscid beetle larvae.

The type of vegetation along a wetland can influence the species composition within the wetland. A laboratory experiment demonstrated that leaves of an introduced shrub (*Multiflora* rose) were less nutritious to *Leptophlebia cupida* than leaves from some kinds of native vegetation, but either increased or had no effect on the stonefly *Soyedina carolinensi (*Sweeny 1993).

Effects on Abundance, Density, Biomass

Vegetation has been demonstrated to support a greater abundance and/or biomass of invertebrates than open water or bottom habitat in a wide range of wetlands, for example: Florida Everglades, Kissimmee River (Florida) wetlands, flatwoods marshes, cypress domes, Southeastern and Pennsylvania beaver pond wetlands, New England riparian sedge meadows, northern prairie marshes, temporarily flooded wetlands in Missouri, seasonal and semipermanent California wetlands, tidal freshwater wetlands, Gulf Coast bayous, and Great Lakes coastal wetlands (Batzer et al. 1999).

The larger growths of epiphytic algae on submerged vegetation may partly explain this greater abundance of macroinvertebrates. In particular, submerged aquatic vegetation with finely divided or thin, submerged leaves and large surface area per unit support higher densities and biomass of macroinvertebrates than emergent vegetation (Parsons and Matthews 1995, Olson et al. 1995). In a Wisconsin lake, densities of *Amnicola limnosa* and *Gyraulus parvus* were 162 and 48 times greater in *Ceratophyllum demersum* beds than in vegetation-free areas (Beckett *et al.* 1992). Areas of the Chesapeake Bay containing the introduced submersed plant, *Hydrilla verticillata*, had greater densities of invertebrates (Posey et al. 1993).

However, densities of invertebrates are not always greater in denser stands of wetland vegetation. In a created Florida marsh, dipteran abundance was greater in unvegetated areas (Streever et al. 1995). Dense stands of vegetation can harbor some predatory fish, and especially in the case of floating-leaved plants, can be associated with reduction in dissolved oxygen. Thus, consideration should be given when monitoring mitigation wetlands to include not only measures of plant stem density, but measures of invertebrate community composition (e.g., Sewell and Higgins 1991, Garono and Kooser 1994).

5.8 Thermal Alteration

Processes

Although lethally hot temperatures are seldom encountered by entire invertebrate communities, the growth and emergence of many wetland insects is closely tied to temperature. In northern climates, freezing can affect invertebrate communities both directly and indirectly. Some invertebrates avoid being frozen by migrating to microhabitats that do not freeze completely, but which often have low concentrations of dissolved oxygen (Euliss et al. 1999). Others are physiologically tolerant of freezing, at least for short periods (Block 1991). Ice also can cause lasting physical and chemical alteration of wetlands. Less dramatic temperature differences between low-elevation wetlands and higher-elevation wetlands can influence invertebrate richness and species composition, so should be factored into monitoring plans (Land & Reymond 1993).

When development encroaches on wetlands, excavations and fills often change the proportion of groundwater vs. surface water inputs to wetlands, and the amount and timing of these inputs. Such changes can alter wetland thermal regimes, and consequently invertebrates (Williams 1991), because each type of water source has characteristic thermal properties – groundwater usually being cooler in summer and warmer in winter, compared with surface runoff. Also, removal of shading vegetation (Section 5.7) affects wetland thermal regimes, as does the direct discharge of heated effluents.

Effects on Richness, Abundance, Density, Growth

Warmer springs tend to have fewer aquatic insect taxa than springs closer to ambient air temperature (Pritchard 1991). Heated effluents also generally decrease the abundance and diversity of macroinvertebrates (Robinson and Craven 1993, Payne 1991). However, in a pair of experimental channels in Ontario, heated vs. unheated channels, after two years, differed little with regard to total invertebrate density, although densities of dipterans were slightly less in the heated channel (Hogg *et al.* 1992). Growth rates of midges (Chironomini, Tanytarsini, Orthocladiinae) in a Georgia blackwater swamp reached a maximum at 21-24°C (Hauer and Benke 1991). Birth rates of the amphipod *Hyalella azteca*, declined when summer temperatures in a Florida lake reached 25-34°C (Edwards and Cowell 1992).

Effects on Species Composition

By helping species at the northern limits of their range to successfully overwinter, warmer winter temperatures can facilitate the northerly range expansion of such peripheral species. For example, the heated discharge of a power plant was critical for the overwinter survival of the Asiatic clam, *Corbicula fluminea* (French and Schloesser 1991). Some species (*e.g.*, the stonefly, *Nemoura trispinosa*, and the caddisfly, *Lepidostoma vernale*) may have mechanisms that enable them to compensate for small changes in thermal regimes, whereas other groups (*e.g.*, Orthocladinae and some other midges) may be more sensitive (Hogg *et al.* 1992).

5.9 Dehydration/ Inundation

Processes

Some of the most dramatic changes to wetland invertebrate communities occur when (a) pools or channels – even seasonal ones -- are introduced into wetlands that seldom or never contained surface water, or (b) wetlands that seldom or never went completely dry are subjected to drought or complete drawdowns. Usually less dramatic are changes to invertebrate communities that occur when slight changes occur in the timing, duration, predictability, and depth of surface water (Eyre 1992, Heicher 1993, Giberson et al. 1992). Permanent water in a wetland can act as a refuge to many species during drought, but also as a disturbance that limits plant growth and development of some invertebrates (Golladay *et al.* 1997). The degree to which desiccation alters invertebrate communities is partly influenced by the severity of the local climate (e.g., maximum and minimum temperature of the wetland sediments during dried-out phase), the ability of plant root systems and detritus to provide moist refuges, and the speed with which water levels fall or rise (Riley & Bookhout 1990).

Wetland water regimes -- including subsurface water table levels -- influence wetland invertebrates by:

(a) altering the amount and pattern of horizontal and vertical habitat space available for colonization;

(b) changing the types of algae and vascular plants that occur, the proportions of these two major energy sources, and the seasons in which they occur (Murkin *et al.* 1991a);

(c) changing the extent of contact between plants and water, thus influencing attachment space, availability of detrital foods (Ross and Murkin 1993, De Szalay *et al.* 1999), shade, and shelter;

(d) altering physical (e.g., temperature, turbidity, oxygen) and chemical regimes (organic carbon, nutrients, metals);

(e) influencing access of predators (Reice 1991, Martin *et al.* 1991, Mallory *et al.* 1994, Johnson *et al.* 1995, Wellborn et al. 1996) as well as the intensity of competition (Wissinger et al. 1999);

(f) in the case of complete desiccation and freezing, being directly lethal to many species (Layzer *et al.* 1993).

Adaptations of major invertebrate groups to desiccation are shown in Table 5.3. Some wetland invertebrates can respond to changing water regime by moving within or among wetlands (Jeffries 1994). An amphipod, *Corophium spinicorne,* and a snail, *Juga plicifera* moved to deeper water within a backdune lake that experienced fluctuating water levels (as much as 2.5 m) in the Oregon Dunes National Monument (Wones and Larson 1991). In a West Virginia stream, the mayfly as *Paraleptophlebia* sp. burrowed toward the water table in response to receding surface water levels (Griffith and Perry 1993). Likewise, in a New York stream the stonefly *Paracapnia opis* survived by following the water table into the hyporheic zone (Delucchi and Peckarsky 1989). In intermittent streams of the Sonoran Desert, some insects escape desiccation by moving up- or downstream (Stanley *et al.* 1994). Also, invertebrates in tidal freshwater wetlands have evolved strategies for dealing with the threat of desiccation (from falling tides) on a daily basis, e.g., Yozzo and Diaz (1999). Taxa that typify many tidal freshwater wetlands include midges and other worm-like taxa, e.g., Tubificidae, Naididae, Enchytraeidae.

Other wetland invertebrates can respond to water level drawdown or drought by laying drought-resistant eggs or by burrowing down to the water table and aestivating. For example, the dessication tolerance of eggs of three mosquitoes -- *Aedes vexans, Aedes trivittatus* and *Psorophora* sp. – allowed them to colonize temporary woodland pools in Michigan (Higgins and Merrit 1999). Resting eggs (also known as epiphia) of some species can survive in complete drawdown conditions for a year or longer. For example, after a complete drawdown that lasted 247 - 346 days, the copepods *Diacyclops haueri*, *Diacyclops crassicaudis brachycercus* and *Acanthocyclops vernalis* emerged from resting eggs within weeks of the return of flooding (Wyngaard *et al.* 1991). After two years the resting eggs of the cladoceran *Diaptomus stagnalis* were able to hatch in laboratory conditions of remoistening (Taylor *et* *al.* 1990). The stonefly *Amphinemura delosa* and the mayfly *Ameletus ludens* survived a complete drawdown in New York streams by having desiccation-resistant eggs (Delucchi and Peckarsky 1989).

Indeed, the density and viability of dormant stages of some invertebrates might be used to determine in advance whether (and how rapidly) the restoration of a drained wetland will restore its functional characteristics (Euliss and Mushet 1999). If sediment samples from a drained or farmed wetland are incubated for about 8 weeks in a moistened laboratory setting, yet fail to produce such hatchings, it might be assumed that degradation has been so severe as to make full functional restoration of the former wetland impractical.

Table 5.3. Adaptations of macroinvertebrates to drawdown or drought (from Smock 1999).

Taxon	Adaptation
Amphipoda	Burrowing, Aestivation
Bivalvia	Burrowing, Aestivation
Caecidotea sp.	Burrowing, Aestivation
Ceratopogonidae	Flight
Midges	Flight
Coleoptera	Flight, Aestivation
Crangonyx sp.	Burrowing, Aestivation
Diaptomus stagnalis	Resting stage
Gastropoda	Burrowing, Aestivation
Isopoda	Burrowing, Aestivation
Ostracoda	Resting stage
Paratendipes sp.	Flight
Polypedilum sp.	Flight
Sphaeriidae	Burrowing, Aestivation
Tabanidae	Flight

Effects on Species Richness

Wetlands with a wide variety of hydrologic zones often have a wide variety of vegetation types, and this in turn can support a wider variety of invertebrates (Kirkman and Sharitz 1994, Williams et al. 1996). Drought or complete drawdown reduces invertebrate richness in many wetlands, and the effect may be temporary or long-term, i.e., still noticeable after wetter conditions return. In particular, the richness of midges diminishes in wetlands following drought (Hershey et al. 1999).

When water is restored to drained former wetlands, invertebrate richness increases during the first few years following restoration (Nilsson and Danell 1981, Hemesath 1991). Richness also tends to increase when water levels are raised in existing wetlands. Similarly, richness (at least of midges) tends to be greater in wetlands having longer durations of standing water during the growing season, whether isolated (Nelson and Butler 1987) or part of a floodplain (Smock 1999). This is partly because wetlands with longer hydroperiods generally are deeper, larger, and more likely to contain submersed and floating-leaved plants that diversify the range of habitats available. Also, wetlands with longer durations of flooding are less likely to experience deep freezing of sediments and types of human activities (e.g., soil compaction, cultivation) that sometimes reduce habitat quality for invertebrates (Swanson et al. 1974). In Wisconsin, only 4 taxa were found in short duration ponds whereas 65 were found in long duration ponds; richness of diving beetles and other predators accounted for much of the increase (Schneider and Frost 1996, Schneider 1999). However, short duration ponds sometimes support more rare species than do longer duration ponds (Collinson et al. 1995). In temporary pools of Colorado, macroinvertebrate richness increased with increasing number of days that seasonal wetlands contained standing water according to the following equation (Wisinger *et al.* 1999):

$$y = 2.52\ e^{(.0232x)}$$

Where x = number of days with standing water; y = number of species present in the wetland

As water permanence increased among wetlands in Grand Teton and Yellowstone parks, food webs became more complex and taxa numbers increased (Duffy 1999). Temporary subalpine wetlands contained 10 species, with the trophic levels culminating in the predaceous beetles *Liodessus* sp. and *Oreodytes* sp.. Seasonal wetlands contained 13 species with the top predator being the longer-lived dragonfly *Lestes unguicalatus*. Semipermanent wetlands supported 14 taxa. Use of emergence traps in 19 Seattle-area wetlands also yielded more taxa from permanently flooded than seasonally flooded wetlands (Ludwa & Richter 2000).

However, after an initial year of continual flooding the invertebrate richness in some inundated wetlands declines. For example, flooding of Manitoba marshes at first increased the variety of both nektonic and benthic invertebrates in vegetation, although not in open water (Murkin et al. 1991, 1992). A short time later, richness of benthic taxa declined (Murkin and Kadlec 1986b). This is perhaps typical. Especially when inundation persists for years with little fluctuation in water level, sediments often become anoxic and light deficits caused by algal blooms can reduce the amount and variety of aquatic plants available as invertebrate habitats, thus reducing invertebrate richness (Neckles et al. 1990).

Effects on Species Composition

A searchable species database on hydroperiod relationships of North American wetland taxa has recently been compiled and is accessible via the internet (Adamus and Gonyaw 2000), so discussion of that topic here will be limited.

Species composition can indicate how long and in what seasons a wetland has contained surface water. This requires that each species found at a wetland first be classified as to its hydrological requirements -- a relatively simple procedure using life history categories such as defined by Hartland-Rowe (1966); McLachlan (1970, 1975, 1985); Wiggins et al. (1980); Jeffries (1989); Eyre et al. (1991); and Batzer and Wissinger (1996). The usefulness of species composition for inferring hydrologic conditions, at least of prairie wetlands, has been demonstrated with midges (Euliss et al. 1993), water beetles (Hanson and Swanson 1989), and macroinvertebrates generally (Neckles et al. 1990, Bataille and Baldassarre 1993).

In general, wetlands can cautiously be deduced to be of greater hydrologic permanence when they contain a higher density and richness of longer-lived and/or relatively immobile species (e.g., snails, mollusks, amphipods, worms, leeches, crayfish), as compared with short-lived species (e.g., anostracans, conchostracans), species that survive the winter as drought-resistant eggs (e.g., *Daphnia*), and/or species that are relatively mobile (e.g., midges, some water beetles and bugs). Drought and drawdown renders the less mobile species more vulnerable to predation, as well as causing their direct loss due to desiccation and related factors (e.g., Stanley *et al.* 1994). Drought also seems to provide competitive advantage to many non-insect invertebrates, and during drought that component can increase at the expense of the insect component of the invertebrate community (Hershey et al. 1999). In a particular seasonal wetland, a dominance of taxa that reputedly survive drawdown by resisting desiccation (e.g., with resistant eggs) rather than by flying away, might suggest that the dry-season microclimate of that seasonal wetland is less harsh than that of wetlands where aerial dispersers dominate (Wissinger 1999). Dispersal characteristics of many taxa have been cataloged in the "Pond-
internet database of invertebrate life history characteristics:
http://www.ent3.orst.edu/PondFX/pondlife_main.htm

Among prairie pothole wetlands, a recent shift to greater hydrologic permanence is suggested by a shift from herbivorous to detrivorous species of macroinvertebrates, and in a shift from open-water forms (e.g., zooplankton, water striders) to forms that characteristically dwell in vegetation (e.g., some mayflies) (Murkin and Kadlec 1986, Murkin et al. 1991). In particular, densities of non-predatory midges (Chironomidae) increase greatly during the first year after flooding, and within this family, species characterized by the greatest tolerance for low oxygen levels increase the most (Murkin and Kadlec 1986b). Densities of swimming (nektonic) and bottom-dwelling (benthic) predatory invertebrates do not increase with flooding as much as do numbers of nektonic and benthic herbivores and detritivores. Predatory species can even decrease after flooding (Murkin et al. 1991), and they often increase as drought or drawdown progresses.

In Minnesota marshes during dry years, molluscs, rotifers and cladocerans were more abundant than in wet years. In wet years, midges, ceratopogonids, copepods, and ostracods were more abundant than in dry years. Stratiomyid flies, water beetles, and craneflies showed no obvious relationship to prior drought (Hershey *et al.* 1999). Flooding of a Manitoba marsh to 1 meter above normal levels led to an increase in the Trichopteran *Agraylea multipuntata*, most likely due to increased algal production and a large increase in decaying macrophyte tissue (Ross and Murkin 1993). In a temporary wetland in California, populations of *Chironomus stigmaterus* expanded during wet years due to increased availability of detritus. Predatory beetles then responded to these larger prey populations (De Szalay *et al.* 1999).

In a year-long comparison of three Manitoba pothole wetlands with different hydroperiods, the permanent wetland was dominated by cladocerans, the semipermanent wetland by ostracods, and the seasonal wetland by copepods (Bataille and Baldassarre 1993). Considering just the emerging aquatic insect component, the permanent wetland was dominated by midges; the semipermanent wetland by water beetles (early season) and midges and other fly species (mid- and late-season); and the seasonal wetland by midges (mid-season) and other fly species (late season). Among temporary pools in

Missouri floodplains, the pools with extended flooding had greater densities of the water beetle *Berosus;* the midges *Dicrotendipes*, *Endochironomus*, *Polypedilum*, and several Orthocladinae; and more copepods and oligochaetes, whereas less permanent pools had more of the Chironomini midges and the midge *Chironomus,* the dipteran *Palpomyi,* and ostracods (Corti *et al.* 1997). In Everglades sloughs, the crayfish *Procambarus alleni* was found mostly in sloughs with short hydroperiod (approximately 168 days) whereas *P. fallax* was found in sloughs with longer hydroperiods (Hendrix and Loftus 2000). Mosquitoes generally are more abundant in temporarily inundated pools, as well as in permanent pools that remain shallower than about 60 cm during the growing season (Batzer and Resh 1992b, Neckles *et al.* 1990). In the Seattle area, richness of emerging mayflies, and insect richness generally, was greater in perennially flooded wetlands than in seasonal wetlands (Ludwa 1994, Ludwa & Richter 2000). Lists of species that have been found in various types of seasonally or temporarily inundated wetlands throughout North America are provided by Batzer et al. (1999).

Among wetlands that are not permanently flooded, the duration and timing of seasonal flooding strongly influences species composition (Schneider & Frost 1996, Wissinger 1999). In a Manitoba marsh, when water persisted only for a few months during the summer (as opposed to the entire growing season), densities were greater of cladocerans, midges and ostracods, whereas diving beetles, corixids, ceratopogonids and ephydrid flies were present in about equal densities regardless of hydroperiod (Neckles *et al.* 1990). In California, marshes flooded in early September had higher winter populations of *Eogammarus confervicolus* and *Berosus ingeminatus* than those flooded in late October. Flooding in September and maintaining water at 40 cm depth, or flooding in October and maintaining water depth of 20 cm, favored the midge *Chironomus stigmaterus* (Batzer et al. 1993). This midge became even more abundant if wetlands were flooded in August (Batzer *et al.* 1997).

Situations sometimes occur where wetland water regime has little affect on species composition (e.g., Neckles et al. 1990). This seems to be the case when the invertebrate fauna is dominated by taxa that characteristically overwinter as adults or larvae, e.g., species of Dytiscidae, Corixidae, Ceratopogonidae, Ephydridae, and some Chironomidae. For example, a survey of five isolated, temporary pools in New York found the “immobile” clams *Pisidium casertanum, Sphaerium occidentale,* and *Caecidotea racovitzai* in three of the pools (Batzer and Sion 1999). Caution also is required in interpreting species composition data because some species with supposedly minimal dispersal abilities are frequently carried passively into reflooded areas by mobile waterbirds (Swanson 1984).

Effects on Abundance, Density, Biomass

Flooding generally increases invertebrate densities as well as richness in wetlands, but perhaps only for about a year after initiation of flooding. For example, flooding of Manitoba marshes containing cat-tail, hardstem bulrush, and common reed to a level 1 m above normal caused a major year-long increase in numbers of nektonic invertebrates in both vegetated and open water areas. Densities of benthic invertebrates increased in flooded vegetation but not in open areas. Biomass of nektonic invertebrates increased only in the vegetated areas (Murkin et al. 1991). On a year-round basis, invertebrate biomass and production in prairie pothole wetlands is probably greatest in semipermanent wetlands (Duffy and Birkelo 1993; Nelson 1989, 1993; Bataille and Baldassarre 1993), but sometimes can reach greater seasonal peaks in temporary and permanent wetlands. In Everglades sloughs, macroinvertebrate densities were three to five times greater in long-duration flooded sloughs (which maintained a water level above 20 cm) compared to short hydroperiod sloughs (levels dropped below 10 cm five times during seven years)(Loftus *et al.* 1990). In Virginia, a floodplain pond that was flooded for 9 months annually had higher absolute abundance, biomass, and production of invertebrates than one that was flooded for much shorter periods (Gladden and Smock 1990). Deeper ponds (>60 cm depth) in California and the Great Plains have greater macroinvertebrate density and lower mosquito abundance than shallower ponds (Batzer and Resh 1992b, Neckles *et al.* 1990). Also, annual reproductive effort can be affected by shortened or lengthened periods of standing water. Populations of the mussel *Musculium partumeium* in permanently flooded Minnesota ponds were able to produce two generations per year, whereas ponds that had seasonal (autumn) or complete drawdown produced only one generation per year (Hornbach *et al.* 1991).

Increasing the duration or depth of standing water does not always increase the density of invertebrates. In Florida Everglades cypress pools, total density of invertebrates may be similar for pools with permanent (perennial) water and pools that dry up for at least a month annually (Leslie *et al.* 1999). In temporary pools in Missouri, invertebrate density is inversely related to water depth, most likely due to interactions with season and reduced algal abundance (Magee *et al.* 1999). In those temporary pools, invertebrate abundance peaked in May and June when water depth was 10-20 cm, and was lowest in the fall when depth was at 60 cm.

Aside from duration and depth of inundation, wetland invertebrate densities can be decimated by rapid water level *fluctuations*, especially when those are more frequent and severe than historically encountered in the wetland. For example, Missouri floodplain pools that experience water level fluctuations at extreme frequency and amplitude tend to have lower invertebrate density (Magee *et al.* 1993). Repeated exposure to desiccation in a short period of time can lead to a marked reduction in invertebrate density. In an Arizona stream that experienced twelve flash floods between August and December of a single year, densities of all invertebrates were reduced from 75 to 100% (Boulton *et al.* 1992). In particular, water spiders, midges, *Probezzia*, Helicopsychidae, *Leptohyphes*, and *Physella* numbers declined whereas oligochaete populations appeared to be unaffected, perhaps due to rapid colonization from upstream source populations.

In contrast, some taxa appear quite resilient to periodic spates. In a British Columbia river, populations of the mayflies *Rhithrogena* and *Baetis*, as well as the caddisfly *Hydropsyche*, survived flows that increased rapidly during flooding from 500 m^3/sec to 6500 m^3/sec (Rempel *et al.* 1999). Survival was assisted by wetlands along the stream that provided temporary refuge from the severe currents, and were a probable source of colonizing individuals immediately after the flood subsided. In an Oklahoma intermittent stream where spring and fall floods reduced invertebrate densities 90%, the mayflies *Caenis* sp., *Leptophlebia* sp. and *Baetis* sp. were especially resilient and midges were less so (Miller and Golladay 1996). When such catastrophic events happen, the apparent “survivors” often are not actual survivors but species with great dispersal capabilities that are able to colonize immediately afterwards from habitats that were less affected by the catastrophe. The pool of available colonizers and the speed at which they recolonize a disturbed area depends partly on the season when severe flooding or drought occurs (Grimm and Fisher 1989). Maintaining minimum water levels (and in streams, flow rates) can increase invertebrate densities, at least during the short term and in the part of a wetland that is not permanently inundated (Weisberg *et al.* 1990, Janicki *et al.* 1990, Troelstrup and Hergenrader 1990).

5.10 Effects of Other Human Influences

Invertebrates such as molluscs that complete their entire life cycle in a single wetland are especially vulnerable to disturbances to (and pollution of) their home wetland, because of their limited ability to escape. In contrast, the dominance of dipterans and other mobile taxa in disturbed wetlands may be explained partly by the short generation time of many of these taxa, and their ability to disperse widely.   However, even species that can disperse aerially may suffer the effects of a regional wetland resource becoming fragmented (i.e., increased distances among suitable wetlands due to wetland loss or degradation). Coincident with watershed development, these characteristic dispersers increasingly encounter factors (e.g., vehicles, pesticides, scarcity of protective cover) during their interwetland movements that increase the risks of interwetland movement, and consequently may suffer reduced survival. At 25 montane wetlands in Switzerland, wetland area and proximity to other wetlands positively influenced the number of specialist butterflies (Wettstein and Schmid 1999). In the heavily agricultural Red River Valley of North Dakota, light-trapping indicated that both the richness of mayflies, and the abundance of the Caenidae mayflies and the Hydropsychid caddisflies, were significantly related to the extent of riverine wetlands (Anderson and Vrondacek 1999). Also influenced by the extent of riverine wetlands were Scarabaeidae beetles, heptageniid mayflies, hydroptilid caddisflies, and ceratopogonid dipterans. Seasonal depressional wetlands were significantly associated with abundance of the water bug *Callicorixa* and ichneumonid wasps. Temporarily inundated depressional wetlands were significantly associated with mayfly richness, as well as abundance of craneflies and hydroptilid caddisflies (Anderson and Vrondacek 1999). These relationships were true during each of two consecutive years.

Habitat fragmentation at regional and local scales can also involve an undesirable homogenization of wetland hydrologic variability. As stated by Wissinger and Gallagher (1999):

> The loss of permanent [water] habitats from a complex [of wetlands] should reduce the pool of cyclic colonizers that seasonally invade temporary habitats, thus reducing diversity in those temporary habitats. Conversely, the loss of temporary habitats should reduce diversity in permanent habitats if such habitats serve as seasonal refugia for species that otherwise would be driven extinct by predators. The loss of temporary habitats should also reduce the rate and sequence of recolonization after drought or other disturbances because they serve as a local source of disturbance-adapted, opportunistic species. For cyclic colonizers, permanent and temporary habitats should act alternatively in different seasons as sources and sinks, depending on whether a species is more vulnerable to drought or to vertebrate predation. The presence of both habitats may be critical for the long-term viability of populations…

The ability of invertebrate habitat in constructed or restored wetlands to compensate for loss of invertebrate habitat in unaltered wetlands, and thus reduce the hazards to dispersers that otherwise would be exacerbated, remains uncertain. Much depends on project design (especially soil treatments and hydrology) and location or the compensatory wetland to other wetlands. Data from 10 natural and 10 constructed marshes in Florida indicated no significant difference in densities of 20 major dipteran taxa (Streever et al. 1996, Evans et al. 1999). A comparison of some recently restored and natural wetlands in New York found that invertebrate communities in the restored wetlands differed initially from natural control sites, but after 3 years were mostly similar (Brown 1995, Brown *et al.* 1997). Another study, in the upper Midwest (Beaver *et al.* 1999), compared constructed wetlands with temporary wetlands and both altered and unaltered wetlands with more permanent hydroperiods. Although species composition varied among the types, the differences in species richness were not statistically significant.   Rotifers accounted for 79% of total zooplankton abundance within the constructed wetlands and were much less dominant in the non-impacted and temporary wetlands. In contrast, other zooplankton (cladocerans and copepods) had low densities in the constructed and impacted wetlands and wer more abundant in the non-impacted and temporary wetlands.

The success of using wetland restoration and creation to mitigate the loss of invertebrate functions depends very importantly on proximity of the compensatory wetland to the altered wetland. This needs thoughtful consideration in wetland banking programs because often these programs, for political and economic reasons, maintain broad flexibility in choosing locations for mitigation.

Also, on a number of occasions humans have wittingly or unwittingly introduced non-native fish and invertebrates to wetlands. Native invertebrate communities seem ill-adapted to compete with or avoid these alien species, but data on community-wide, long term effects are mostly lacking. Studies of the northern Everglades have discovered some macroinvertebrate colonists from Central and South America (Rader 1994). Zebra mussel (*Dreissena polymorpha*) has invaded many aquatic systems throughout North America (d'Itri 1997). This species can totally carpet substrates, displacing native mussels (Tucker 1993, 1994), some midges (Chironomini and Tanypodinae), snails *Polycentropus* sp., *Physella* sp., *Pleruocera* sp., and the caddisfly *Oecetis* sp., while having minimal or positive effect on amphipods and flatworms (Wisenden and Bailey 1995). They may also concentrate contaminants, making them more available to invertebrate food chains (Bruner et al. 1994). Ironically, the rapid spread of zebra mussels may have been partly attributable to the preceding decline of native mussels as a result of pollution and habitat alteration (Roberts 1990, Nalepa and Schloesser 1991b, 1993, Hebert *et al.* 1991, Mackie 1991, Haag *et al.* 1993, Whittier et al. 1995). Because unionid mussels in rivers are relatively immobile and have long life spans (often over 10 years), they are particularly susceptible to disruptions from introduced mussels as well as from impoundments and channelization (Mehlhop and Vaughn 1994). Riverine wetlands with higher alkalinity tend to be more susceptible to invasions by zebra mussels (Whittier *et al.* 1995), although wetlands generally tend to be less suitable than other habitats for zebra mussels (Griffiths *et al.* 1991). Indeed, wetlands along rivers might serve as refuges for native mussels otherwise impacted by zebra mussel expansion (Tucker and Atwood 1995).

In boreal regions, wetlands that contain fish (even native fish) had fewer macroinvertebrates than wetlands without fish (Mallory *et al.* 1994). Stocking or accidental release of fish into wetlands unaccustomed to harboring fish can have a major impact on the invertebrate communities (Johnson *et al.* 1995, Martin *et al.* 1991).

5.11 Wetland Monitoring

Spatial and Temporal Variability

Choice of appropriate sample sizes depends on measured variation in the target taxa and metrics. Such coefficients of variation were calculated from previous invertebrate studies in prairie pothole wetlands (Adamus 1996), and are summarized from various aquatic studies elsewhere at:
http://www.im.nbs.gov/powcase/powvariation.html

One major source of sampling variation is the variation among habitats within a wetland. Information on within-wetland invertebrate variability is presented (at least qualitatively) in the book edited by Batzer et al. (1999) and is available (at a minimum) for: Florida Everglades (Rader 1994, 1999), Florida flatwoods wetlands, southern forested floodplain wetlands, forested limesink wetlands of Georgia (Golladay et al. 1997), bay wetlands of the Carolinas, beaver ponds in the Southeast, beaver ponds and constructed marshes in Pennsylvania, Canadian peatlands, depressional wetlands of Ohio, Lake Michigan wetlands (King and Brazner 1999), prairie potholes (Euliss and Mushet 1999), lacustrine wetlands (Murkin *et al.* 1991) of Manitoba, ricelands and seasonal and semipermanent wetlands of California (de Szallay and Resh 1997), and created wetlands (Cooper and Anderson 1996) and High Plains wetlands of Wyoming. Some other significant sources of variation include geographic region, season, and daily weather conditions (Anderson and Vondracek 1999).

Techniques and Equipment

The growing number of sampler types used for sampling invertebrates are described in Adamus and Brandt 1990, Cuffney et al. 1993a,b, Bookhout 1994, Adamus 1996, Merritt and Cummins 1996, and other sources. They include, for example:

- artificial substrates (Magee *et al.* 1993, Benoit et al. 1998)
- sweep nets, to collect planktonic and epiphytic invertebrates (Cheal et al. 1993, Batzer *et al.* 1993)
- throw traps (Turner & Trexler 1997)
- light traps (Anderson and Vondracek 1999)
- bilge-pump samplers (Batzer *et al.* 1993)
- stovepipe samplers (Schwartz et al. 1994)
- activity traps (Murkin *et al.* 1991)
- Ekman dredge (Blomqvist 1990)
- coring devices (Leslie et al. 1997)

All of these have drawbacks that make some taxa less likely to be captured than others, so a combination of methods should be used whenever possible (Brinkman & Duffy 1996, Turner & Trexler 1997). If a specific type of organism is to be targeted (benthic, planktonic, epiphytic, etc.), sampling should be designed to exclude influences from other community types. As an example, Batzer *et al.* (1993) sampled both benthos and the water column of a marsh. Benthos was sampled using a bilge pump sampler in which the opening of the sampler is planted firmly on the substrate, which was drawn through the device into a sieve. Water column invertebrates were sampled using a 1-mm mesh sweep-net kept 5 cm above the benthos.

Among the more novel (for wetland studies) approaches that have been used in the last decade to sample invertebrates are sticky traps (Nordstrom and Ryan 1996, King and Brazner 1999); resting egg counts (Euliss and Mushet 1999); and the various soil extraction and other methods used to sample earthworms and nematodes in soils and sediments of seasonally dry wetlands (Lenz & Eisenbeis 1998, New 1998) or sub-arctic wetlands (McElligott & Lewis 1994). Special techniques are often required to accurately survey freshwater mussels (e.g., Miller 1991).

Identification of Taxa

Some of the more recent and commonly used keys for North American aquatic invertebrates include Merritt and Cummins (1996), Thorp and Covich (1991), Pennak (1989), and Peckarsky *et al.* (1990). Also, some recent keys address specific groups, e.g., damselflies (Westfall and Minter 1996) and caddisflies (Wiggins 1996). Results are mixed regarding the necessity of identifying invertebrate taxa to species. While generally desirable, this decision clearly depends on the available resources and the objective of the study. Tests of alternative multimetric indices in the Seattle area showed that indices based on identifications of emerging insects only to the family level failed to distinguish an urban land use gradient among wetlands, whereas indices based on genus/species levels were much more sensitive (Ludwa 1994, Ludwa & Richter 2000).

Metrics and Indices for Assessing Impacts to Wetland Invertebrate Communities

Multiple metrics (including indicators, indexes) have been used for assessing the condition of surface waters and are described in Merritt and Cummins (1996) and Karr and Chu (1999). Following are some examples of recent studies that have examined wetlands that span a regional gradient of anthropogenic (generally land use) disturbance. Most of these studies used a multimetric approach to assessing wetland condition.

Montana (Apfelbeck 1998):
Macroinvertebrates were sampled in 80 wetlands statewide. The wetlands belonged to ten classes that were defined by salinity, morphology, hydroperiod, and landscape position. The multimetric approach did not work well for wetlands that lacked open water (partly because of the sampling methods used), or for excessively saline or alkaline wetlands. The approach succeeded in indicating the relative level of impairment (of a wetland's invertebrate communities) but was less useful in diagnosing a cause of the impairment. Impairments were likely related to acidification, sedimentation, excessive enrichment, contamination with heavy metals, and hydroperiod alteration. The following metrics were judged to be too variable to distinguish water quality impacts in the study wetlands: % collector-gatherers, % filterer-collectors, and ratio of Tanytarsini to total Chironomidae. Also, the metric, "% amphipods" was not used because in Montana it was correlated naturally with pH (alkalinity).

The final "core" metrics that were used and combined into the multimetric index were:

number of taxa
percent dominance (cumulative total of %'s of 1, 2, and 5 most dominant taxa)
number of taxa that are stoneflies, mayflies, dragonflies, or caddisflies
total number of individuals per sample
number of taxa that are Crustacea or Mollusca
number of taxa that are leeches, sponges, or clams
mean tolerance values of component species

The range of multimetric values, and of the individual metrics, found in each wetland class is presented in the report.

Washington (Ludwa 1994, Ludwa & Richter 2000):
Aquatic insects were sampled in 19 Seattle-area wetlands during 3 nonconsecutive years. The wetlands were exposed to stormwater and urban runoff to varying degrees. Insects were sampled solely with emergence traps, mostly checked on a monthly basis. The metrics found to be most useful for distinguishing developed from undeveloped watersheds were:

Taxa richness
Richness of Ephemeroptera + Plecoptera + Odonata + Trichoptera (EPOT) taxa
Richness of Tanytarsini taxa
Richness of Chironomini taxa
Richness of Tanypodini taxa
Percent of individuals as EPOT
Percent of individuals as Tanytarsini
Percent of individuals as Chironomini
Percent of individuals as Tanypodini
Scraper and/or piercer taxa presence
Shredder taxa presence
Collector taxa presence
Presence of *Thienemanniella*
Presence of *Endochironomus nigricans*
Presence of *Parachironomus*
Presence of *Polypedilum*
Presence of *Ablabesmyia*
Presence of *Aspectrotanypus algens*
Presence of *Paramerina smithae*
Presence of *Psectrotanypus dyari*

Presence of *Zavrelimyia thryptica*
Presence of *Tanytarsus*

Minnesota (Gernes and Helgen 1999):
Invertebrates were sampled in 27 depressional wetlands in the Central Hardwood Forest ecoregion, using standardized dipnetting and bottle traps. Ten metrics were tested (see the report for full details):

% Corixidae + % Coleoptera
% Erpobdella leeches
sum of %'s of 3 most abundant taxa
# of genera of caddisflies, mayflies, and sphaerid clams
# of genera of midges
# of intolerant taxa (*Leucorrhinia, Libellula, Tanytarsus, Procladius, Triaenodes, Oecetis*)
# of leech genera (Hirudinidae)
# of genera of dragonflies and damselflies
# of snail taxa (mostly to species)
total richness

The metrics based on leeches and snails were least effective in discerning the land use gradient. The most discerning metrics were the # of intolerant taxa, the # of genera of dragonflies and damselflies, and # of genera of caddisflies, mayflies, and sphaerid clams. Presence of fish in some of the wetlands apparently did not confound the use of the multimetric index to discern the disturbance gradient.

Another study in Minnesota sampled invertebrates in over 100 wetlands belonging to 8 types, with sites representing each type selected to span a gradient of land cover. Results of testing over a dozen candidate metrics for positive or negative correlations with land cover types at various distances around each wetland are reported at:
http://www.hort.agri.umn.edu/mnwet/begin.htm

Prairie wetlands (Anderson and Vondracek 1999):
Aquatic insects were sampled in light traps placed near 126 wetlands during 2 years. Surrounding land cover was mainly grassland or cropland. In general, the effects of surrounding land cover were statistically overshadowed by geographic region and daily weather. However, the following metrics showed some usefulness as indicators of surrounding land cover condition, especially when data were separated by ecoregion:

abundance of aquatic insects
abundance of Caenidae mayflies
abundance of Scarabidae beetles
abundance of Lepidoptera moths
mayfly richness

Lake Huron wetlands (Burton et al. 1999):
In a comparison of three relatively pristine and three impacted Lake Huron wetlands, 24 potential metrics were tested for their effectiveness at discriminating between altered habitats. Fourteen of the metrics appeared to respond reliably to disturbance across a variety of wetland vegetation types, although the sensitivity of the metrics differed (Table 5.4).

Table 5.4. Macroinvertebrate community metrics tested in 3 impacted and 3 unimpacted Lake Huron wetlands (Burton *et al.* 1999).

Metrics that displayed the same response over all vegetation types are bolded.

METRIC	Direction of response with increasing disturbance in each vegetation type[1] OS	IS	TY	WM	ALL
# of Crustacea + Mollusca genera	**D**	**D**	**D**	**D**	**D**
# of Ephemeroptera + Trichoptera genera	N	D	N	N	N
# of Ephemeroptera genera	N	D	N	N	N
# of Odonata genera	**D**	**D**	**D**	**D**	**D**
# of Trichoptera genera	I	D	N	N	N
Total # of taxa	**D**	**D**	**D**	**D**	**D**
Total # of genera	**D**	**D**	**D**	**D**	**D**
Total # of families	D	D	N	N	D
% Amphipoda	N	N	I	N	N
% Midges	I	I	D	D	N
% Crustacea + Mollusca	D	D	N	N	N
% Ephemeroptera	D	I	D	D	N
% Gastropoda	**D**	**D**	**D**	**D**	**D**
% Isopoda	D	D	N	I	N
% Odonata	**D**	**D**	**D**	**D**	**D**
% Spheariidae	**D**	**D**	**D**	**D**	**D**
% Tanytarsini	I	I	N	D	N
% Trichoptera	I	D	I	D	N
% Diptera	I	I	D	N	N
% Crustacea	N	D	N	I	N
Evenness (J')	**D**	**D**	**D**	**D**	**D**
Shannon Index (H')	**D**	**D**	**D**	**D**	**D**
Simpson Index (D)	**I**	**I**	**I**	**I**	**I**

[1] OS = Outer *Scirpus*, IS = Inner *Scirpus*/Pickerelweed, TY = *Typha*, WM = Wet meadow, ALL = All sampling stations combined.

Reference Values

From a review of regional literature, one might begin to quantify the "expected" range of variation – both natural and unnatural – in density and species richness, and also to specify taxa that characterize least-altered (reference) wetlands of each type in the region. Knowing such reference conditions provides a basis for comparison (benchmark) useful for interpreting data collected in future studies. However, such a compilation of values and taxa would be confounded by the lack of commonality of field methods, equipment, units of measurement, and schemes for classifying habitats within wetlands and the wetlands themselves, as well as by unknown biases in selection of study sites. Nonetheless, much useful information of this type is presented in the book edited by Batzer et al. (1999) and from other sources, and is available (at a minimum) for: Florida Everglades, forested limesink wetlands of Georgia, snowmelt ponds in Wisconsin, beaver ponds and constructed marshes in Pennsylvania, riparian sedge meadows in Maine, pitcher plant bogs of eastern Canada, depressional wetlands of Ohio, prairie potholes, riverine and prairie wetlands of Minnesota, lacustrine wetlands and prairie wetlands (Wrubleski 1999) of Manitoba, High Plains wetlands of Wyoming, playas of the southern

High Plains, wetlands of Yellowstone and Grand Teton parks, tidal freshwater marshes of Virginia, seasonal and semipermanent wetlands of California, and urban wetlands of western Washington .

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Section 6. Fish

6.1 Use as Indicators

This section addresses fish that are closely associated with non-tidal wetlands. Much, perhaps most, of the literature is from studies of lacustrine fringe wetlands (e.g., littoral vegetation). For a general discussion of the topic based on pre-1990 scientific information, and for discussion of advantages and disadvantages of using fish as indicators of wetland integrity, readers may refer to Adamus and Brandt (1990).

In contrast to fish monitoring in streams (e.g., Bramblett & Fausch 1991), there have been few attempts to develop fish IBIs (indices of biotic integrity) specifically in non-tidal wetlands of North America. In Minnesota, wetland and stream fish were used together to represent the condition of landscapes that contained a large wetland/ riparian component (Galatowitsch et al. 1998, Mensing et al. 1998). Sampling 15 wetlands belonging to each of 7 wetland types, the investigators found positive or negative correlations of the following metrics with a site disturbance score and/or various land cover types measured within 500, 1000, and 2500 m of each wetland:

- In small-sized river floodplains: fish species richness, proportion of Cyprinids
- In medium-sized river floodplains: fish species richness
- In large river floodplains: the proportion of piscivores, total abundance of fish, proportion of Catostomids
- In non-calcareous littoral wetlands: total fish abundance
- In calcareous wetlands: species richness, total abundance, proportion of Cyprinids, number of sunfish species
- In forest glacial marshes and prairie glacial marshes: total abundance, richness

Results are reported by wetland type at: http://www.hort.agri.umn.edu/mnwet/

Attempts to apply a fish IBI to lacustrine wetlands in 60 Florida lakes met with mixed success (Schultz et al. 1999). Fish IBI scores increased with increasing nutrients (lake trophic status) and lake surface area. Metrics used in the IBI were: total fish, native fish, *Lepomis*, piscivores, generalists, insectivores, and intolerant and tolerant species. Attempts to develop and apply a fish IBI to isolated wetlands near Lake Michigan are described by Simon (1998a,b).

The effects on North American fish of many human-associated factors are summarized by Miller et al. (1989) and Hughes and Noss (1992).

6.2 Effects of Enrichment, Eutrophication, Reduced Dissolved Oxygen

As eutrophication increases plant and algal productivity, fish sometimes suffer from reduced levels of dissolved oxygen, and feeding habits also may shift. Biomass and species richness may increase or decrease, depending on the initial state of the wetland and the duration and magnitude of the eutrophication. To some degree, fish families can be grouped according to decreasing susceptibility to oxygen deficiencies: salmonids and coregonids require high levels of dissolved oxygen, whereas cyprinids often tolerate low dissolved oxygen levels (Harper 1992).

In a study of an Everglades wetland, *Gambusia holbrooki*, *Heterandria formosa* and some other small fish were 2 to 3 times more abundant in enriched wetlands than unenriched wetlands (Rader and Richardson 1994) and biomass (standing stock) was greater (Turner et al. 1999). An isolated Florida wetland receiving treated wastewater supported similar diversity and higher abundance of fish, compared with its condition prior to receiving the wastewater (Schwartz et al. 1994). In another wetland receiving advanced secondary treated wastewater in Central Florida, fish populations were similar to those in a wetland not receiving wastewater (Best 1993). A survey of 60 natural lakes in Florida found pygmy killifish, lined topminnow, chain pickerel, and redfin pickerel in lakes with the least phosphorus and/or nitrate (Hoyer and Canfield 1994).

In an Alaskan tundra river system, fertilization with phosphorus nearly doubled the size of young-of-the-year fish, suggesting that phosphorus limitation of algal production had limited the fish previously (Deegan & Peterson 1992).

6.3 Effects of Contaminant Toxicity

Smaller fish may be the first members of a fish community to elicit a response to contaminants (Matuszek *et al.* 1990) due to their high metabolic rate relative to larger fish (Shuter and Post 1990). The body burden of copper, cadmium, and silver in minnows can be a strong indicator of biologically available heavy metals and thus of ecological impact (Birge et al. 2000). The toxicity of copper and zinc to some fish species depends on other chemical characteristics of waters (Munkittrick & Dixon 1992, Welsh *et al.* 1993, Erickson *et al.* 1996), as well as fish behavior (Pourang 1995). For example, presence of 5 mg C/L as dissolved organic matter from a marsh kept copper from binding to the gills of small *Oncorhynchus mykiss*. This occurred due to the complexing of copper with dissolved organic carbon, making the copper unavailable (Hollis *et al.* 1997). Some fish species appear capable of becoming acclimatized to moderately elevated levels of some metals (Klerks & Lentz 1998).

The impact of combined contamination from heavy metals and acidification was investigated in a Canadian beaver pond receiving ore smelting effluent from an abandoned industrial center (Rutherford and Mellow 1994). Besides contributing nickel, copper, aluminum, iron, lead, zinc, arsenic, cadmium and chromium, the effluent acidified the water. Immediately downstream from the source, no living fish were captured during 4 months of seining effort, implying either extreme avoidance of the area by fish or severe mortality of resident fish. Fish captured within 0.1 km upstream of the contamination included *Culaea inconstans, Phoxinus neogaeus, Phoxinus oes,* and *Pimephales promelas*.

Bioaccumulation of mercury in fish is a growing concern in many North American lakes and wetlands. Properties of individual lakes appear more important for determining fish tissue mercury concentrations than do small-scale ecoregional differences. In a survey of 24 Massachusetts lakes relatively unimpacted by mercury, lake trophic state did not influence mercury levels in fish tissues, whereas pH was highly (inversely) correlated with mercury in fish tissue, at least in yellow perch and brown bullhead (Rose *et al.* 1999). Mercury accumulation by fish is especially great during the first 2 years an area is flooded (Kelly et al. 1997, Bodaly & Fudge 1999) but does not always occur (Miles & Fink 1998). Tissue mercury levels typically vary greatly by fish size and trophic level. In 13 Ontario lakes, mercury in tissues of smallmouth bass was positively correlated with crayfish mercury levels (Allard and Stokes 1989). In Florida, mercury found in mosquitofish was lower than in bass and was lower in the wetland interior than in the inflow and outflow channels (Miles & Fink 1998).

Selenium is not directly toxic to fish at usual concentrations, but can become toxic once concentrated in fish food chains, especially in some wetlands that receive effluents from irrigated fields or power plant reservoirs in some regions (Zilberman 1991, Lemly 1996).

Synthetic organics, including pesticides, can accumulate in wetland fish (Cooper 1991), often with adverse effects. In a Canadian wetland receiving oil sand effluent, fish had altered blood chemistry and died within 14 days (Bendell-Young et al. 2000). Two herbicides used to control the invasive wetland plant, purple loosestrife (*Lythrum salicaria*), were not toxic to rainbow trout (Gardner and Grue 1996). Sensitivity of smaller non-game wetland fish was not investigated. Fish exposed to pesticides, PCB's, and other synthetic organics are often more vulnerable to disease, and the literature on this subject was reviewed by Dunier & Siwicki (1993).

6.4 Effects of Acidification

Surveys of literature on effects of acidification on fish, including pH values critical to fish response, are provided by Baker and Christensen (1991), Minns *et al.* (1990), Carline et al. (1992), and others. Acidity can be directly toxic to fish, inhibit reproductive maturation, inhibit spawning behavior, induce emigration, and alter food availability (Baker and Christensen 1991). Acidity induces aluminum toxicity in fish in many lakes and wetlands, although in Florida, soil characteristics made this less problematic (Keller and Crisman 1990). Acidities in the range of 5.0 to 6.0 pH are critical for several species. In an Ontario beaver pond, a pH of 3.7 resulted in complete mortality due to acid-related release of toxic metals from sediments (Rutherford and Mellow 1994). Models for fish response to acidification are provided by Charles (1991). Ten species of the Upper Midwest - four cyprinids, three percids, *Coregonus artedii*, *Percopsis omiscomaycus*, and *Lota lota* - were not caught in waters with pH < 6.0 (Cook and Jager 1991, Cusimano *et al.* 1990). *Perca flavescens* and *Umbra limi* were the most acid-tolerant in the studies of lakes in Michigan and Wisconsin. Other acid-tolerant species included *Culaea inconstans*, *Ictalurus nebulosus*, *Lepomis macrochirus*, and *L. gibbosus* (Cook and Jager 1991). However, factors such as recreational fishing, lake size, and predation confound attempts to attribute fish absence to a particular contaminant (Cook and Jager 1991). Table 6.1 classifies fish by acid sensitivity in waters of upstate New York (Schofield and Driscol 1987 as cited in Charles 1991).

Table 6.1. pH associations of fish of the north branch of the Moose River, New York (from Schofield and Driscol 1987).

Sensitive - found only in waters with pH > 6.0	Intermediate - found in both types of waters	Tolerant - found in waters with pH < 5.0
Blacknose dace	Brook trout	Mudminnow
Redbellied dace	White sucker	Killifish
Common shiner	Creek chub	Brown Bullhead
Smallmouth bass	Pumpkinseed sunfish	Golden Shiner
Rock bass	Finescale dace	Yellow Perch

Among 426 Ontario lakes, cyprinid species declined when pH was below 6.0, and no cyprinid species were found below pH 4.7. Lakes below pH 5 tended to be dominated by *Perca flavenscens, Lepomis gibbosus* and *Ambloplites rupestris.* Above pH 6, lake size (across a range of 10-1585 ha) influenced fish species richness more than pH, or neither pH nor lake size had any discernable effect (Matuszek *et al.* 1990) Regional environmental differences can mediate the impact of water quality characteristics. For example, subtropical Florida lakes had higher numbers of fish species than similar temperate lakes despite having approximately the same pH (Keller and Crisman 1990). In Florida, 11 fish species were found in lakes with a minimum measured pH of 4.3 (Hoyer and Canfield 1994). These included the lined topminnow, Everglades pygmy sunfish, pygmy killifish and redfin pickerel.

6.5 Effects of Salinization

The age structure and growth rate of *Micropterus salmoides* from a brackish marsh and a freshwater oxbow lake system was investigated in south-central Louisiana (Meador and Kelso 1990a, b). Marsh fish exhibited small size and reduced length at age. However, growth rates of older marsh fish equaled or exceeded those of freshwater largemouth bass. Brackish water fish maintained better body condition throughout the year whereas freshwater fish exhibited reduced condition during early spring and fall. Laboratory trials consisting of 120-day exposure of marsh and freshwater largemouth bass to four salinity levels (0, 4, 8, and 12 ppm) indicated a significant decrease in growth rate of freshwater largemouth bass with increasing salinity level up to 8 ppm. No such decrease was observed for marsh largemouth bass. All fish held at 12 ppm stopped feeding within one week and died before the end of the experiment.

In another experiment, juvenile bluegill (*Lepomis macrochirus*) from a freshwater pond in northeastern Mississippi and a brackish bayou in coastal Mississippi were held in a chamber with 0 ppm salinity but given access to chambers containing 0, 2, 4, 6, 8, and 10 ppm salinity (Peterson *et al.* 1993). Fish from neither habitat showed clear preference for any of the salinity options. These data and data from previous studies suggest bluegill are better able to physiologically and behaviorally tolerate elevated salinity relative to other centrarchids, particularly bass (*Micropterus*) (Peterson *et al.* 1993).

A model for assessing effects on wetland fish of major cations from irrigation runoff was tested and described by Dickerson et al. (1996).

6.6 Effects of Sedimentation/Burial

No recent studies specific to North American wetlands or lakes were found.

6.7 Effects of Thermal Alteration

Laboratory experiments and field observations of centrarchid, poeciliid, and cyprinodontid fishes in the Florida Everglades demonstrated that suddenly cooler temperatures (15 °C minimum daily temperature) led to fish behavioral changes that might affect higher trophic levels by reducing prey availability (Frederick and Loftus 1993). Throughout Ontario, lake elevation (a surrogate for temperature and possible absence of predatory fish) had a statistically strong and positive influence on abundances of yellow perch and white sucker (Hinch et al. 1994).

6.8 Effects of Vegetation Removal

Removal of vegetation from within or alongside wetlands affects wetland fish largely by increasing water temperature and susceptibility to predation, and by altering foods and their availability. Woody material is especially important as a source of cover for fish in off-channel wetlands (e.g., oxbows, sloughs) and in lakes (Leitman et al. 1991, Dewey & Jennings 1992, Fausch and Northcote 1992, McIntosh *et al.* 1994), but shoreline trees are often cut to improve landowner views or for firewood (Christensen et al. 1996). Larval fish prefer structurally complex habitats in Mississippi oxbow lakes (Killgore and Miller 1995, Killgore and Baker 1996) and probably in other systems.

In lacustrine fringe wetlands, submerged macrophytes are particularly important. For example, declines in macrophytes (resulting from grass carp introductions, Bain 1993) have been linked to an increase in the proportional abundance of limnetic (open water) fish species (Bettoli *et al.* 1991, Maceina *et al.* 1991, Martin et al. 1992). Macrophyte diversity and coverage also were identified as key determinants of species composition of fish communities in Lake Michigan wetlands (Brazner and Beals 1997). Nonetheless, intentional thinning of extensive macrophyte beds can result in higher growth rates of some age classes of lake fish, presumably due to improved access of fish to invertebrate foods (Olson et al. 1998).

At least for some fish, there is little difference in their propensity to use aquatic beds of native plants as opposed to aquatic beds of non-native, invasive plant species (Conrow et al. 1990, Duffy & Baltz 1998).

6.9 Effects of Turbidity, Shade

Turbidity, partly through its adverse impacts on submersed macrophytes and algae, affects the structure of many wetland fish communities (e.g., Lake Michigan wetlands, Brazner and Beals 1997). Bottom-feeding carp (*Cyprinus carpio*) often are a major contributor to turbidity in shallow lakes and wetlands, due to their large size and regular stirring of sediments as they feed. Turbidity, total phosphorus, and total ammonia concentrations increased predictably with total biomass of carp stocked in experimental enclosures within an Ontario marsh (Lougheed *et al.* (1998) Although carp had no direct effect on zooplankton community structure, zooplankton biomass declined due to increased turbidity and altered nutrient availability.

Shade produced by increased cover of duckweed after the addition of wastewater to a blackwater wetland in Central Florida caused a reduction in *Gambusia* (Smith 1992).

6.10 Effects of Dehydration/Inundation

Spring-fed isolated wetlands in the western United States provide the only habitat for several rare fish (Meffe 1989). Among isolated wetlands of the upper Atlantic Coastal Plain, fish (as expected) inhabit only those wetlands that dry infrequently and/or have elevations similar to nearby permanent waters that periodically serve as a source of colonists (Snodgrass *et al.* 1996). Along the lower Missouri River, twice as many fish species were found in sloughs regularly connected to the main channel as in isolated sloughs that were not. The fish community in the connected sloughs also contained a larger component of native species (Galat et al. 1998). In the northern United States, isolation of small lakes from other water bodies also influences fish species composition (Magnuson et al. 1998).

In riverine systems, the availability of off-channel or in-channel pools that hold water through dry periods and provide calm-water feeding areas during floods is a critical feature in the stability of fish communities in both riverine and non-riverine wetlands (Gelwick 1990, Capone and Kushlan 1991, DeAngelis et al. 1997). Floods that create such areas, by periodically rearranging the geomorphic structure of floodplains and importing large woody debris from normally unflooded areas are essential to maintaining fish habitat in large rivers (Galat et al. 1998, Michener et al. 1998, Shields et al. 2000). Floods also are important because they provide adult fish with temporary access to a rich supply of relatively unexploited floodplain foods (Leitman et al. 1991, Killgore & Baker 1996, Jordan et al. 1998).

Declines in water level, whether drought-related or human-caused, alter community structure of wetland fish, partly by creating greater overlap of resource utilization and increasing predation risk. In a North Dakota stream that experienced 6 years of drought, species richness declined from 23 to 13 species (Kelsch 1994). Low water also increases chances of fish freezing in winter and being subjected to lethal thermal stress in summer. Sustained drawdowns, by temporarily eliminating larval dragonflies and other large invertebrates that normally compete with or prey on larval fish, also can reduce competition among fish that return to wetlands when water levels rise again (Travnichek & Maceina 1994, Jordan et al. 1996). Isolated wetlands that experience frequent near-drought conditions tend to support mainly smaller species, and small individuals of normally larger species (Loftus and Eklund 1994).

6.11 Effects of Other Stressors

Non-native fish have been widely introduced into waters of the United States, both on purpose and by accident. Pollution and alteration of water regimes sometimes accelerate invasion of natural habitats by non-native fish species, which tend to be broadly tolerant. Effects of invaders on native fish communities are usually adverse (Baltz and Moyle 1993), especially when coupled with simultaneous impacts from other factors (Larimore & Bayley 1996, Marschall & Crowder 1996). Consequently, dominance by non-native species is often used as a measure of low biological integrity (Fausch *et al.* 1990, Farr & Ward 1993), although native fish communities may sometimes be impacted by watershed urbanization even in the absence of invasion by non-native species (Weaver and Garman 1994). A survey of the St. Louis River estuary revealed the invasive *Gymnocephalus cernuus* was the fifth most abundant species in areas impacted by human disturbance and the 15th most abundant species in the relatively unimpacted inner marsh. Invasion by this species appeared to be inhibited by the presence of dense wetland vegetation (Brazner *et al.* 1998). Careful sampling of New England lakes indicates that assemblages of minnows have been devastated by introduction of non-native predatory fish (present in 69% of the lakes) and by often-accompanying development of shorelines and watersheds (Whitter et al. 1997).

Like larval amphibians, young fish of some species have been found to be highly sensitive to ultraviolet-B radiation (Ewing et al. 1999).

A diked wetland on Lake Erie was found to support 23 fish species, whereas an undiked wetland nearby supported 40. The five most abundant fish in the diked wetland were *Pomoxis annularis,* *Amieurus nebulosus*, *Amieurus melas*, *Carassius auratus* and *Cyprinus carpio*. Comparatively, the five most abundant fish in the undiked wetland were *Morone americana*, *Dorosoma cepedianum*, *Pomoxis annularis*, *Amieurus nebulosus*, and *Perca flavescens*. Mean summertime lengths of several fish species were significantly greater in the undiked wetlands (Johnson et al. 1997).

Suitably-designed constructed wetlands have been reported as usually supporting a richness and abundance of fish comparable to or greater than natural wetlands (e.g., Langston and Kent 1997, Morrow 1999). However, mean abundance and biomass of *G. holbrooki* and *E. evergladei* in some constructed Florida wetlands differed from natural wetlands there (Streever and Crisman 1993). The capacity of stream fish communities to recover from disturbance is discussed by Meffe & Sheldon (1990), Detenbeck et al. (1992), and Kinsolving & Bain (1993). Wetland fish populations in the Florida Everglades may require up to 1 year to recover from drought (DeAngelis et al. 1997).

6.12 Wetland Monitoring

Spatial and Temporal Variability

In six Wisconsin lakes, annual variation in fish community structure was minor, indicating that each lake was fairly stable in time. In contrast, variation in fish communities among lakes was much greater, a consequence of substrate, macrophyte and depth variations (Benson and Magnuson 1992). Low species richness in one of the lakes was accompanied by overwhelming dominance of a few species (Magnuson and Lathrop 1992). Within a lacustrine fringe wetland of Lake Huron, fish richness and abundance decline 40-70% with distance from the open water edge (Bouchard 1998). In a Texas reservoir, a single habitat type hosted a variety of fish species, depending on the season (Gelwick and Matthews 1990). Fringing wetlands provide spawning and rearing areas in most lakes, and in large lakes may be much more important than their proportionate size alone would suggest (Jude and Pappas 1992). Lake fish communities tend to be the result of multiple structuring factors, events, and processes that produce communities that are nearly unique to each water body (Tonn 1990, Tonn *et al.* 1990). The spatial distribution of fish in Florida wetlands also changes with season (Jordan et al. 1998).

Techniques and Equipment

Fish are captured in wetlands and lakes using a variety of methods. Information on equipment and guidance for sampling fish in streams and/or lakes is provided by Meador et al. (1993), Murphy and Willis (1996), and many reports that pre-date the review period covered by this document. Equipment includes but is not limited to:

- seines (Pierce et al. 1990, Rutherford and Mellow 1994)
- fyke nets (Weaver & Magnuson 1993, Brazner et al. 1998)
- dip nets
- throw trap (Chick et al. 1992, Jordan et al. 1997)
- pop nets (Petering and Johnson 1991, Dewey 1992)
- minnow traps (He & Lodge 1990)
- electroshocking (Dewey 1992)

Although more difficult and potentially dangerous, sampling at night is often much more effective than daytime sampling, and frequently reveals species assemblages and habitat use patterns much different than found during the day.

Very different kinds of gear are needed for sampling fish larvae. This may include plankton tow nets (Conrow *et al.* 1990) and floating light traps (Killgore & Baker 1996)

A variety of biochemical and other methods are of potential use for assessing health of individual fish in relation to toxic chemicals. A review was published by Niemi (1990).

Metrics for Assessing Impacts to Wetland Fish Communities

Information on this topic as well as sampling equipment is reviewed on EPA's wetland biomonitoring web page: http://www.epa.gov/owow/wetlands/bawwg

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Section 7. Amphibians And Reptiles

7.1 Use as Indicators

This section addresses the monitoring of turtles, frogs, toads, salamanders, newts, snakes, crocodilians, and lizards that occur in wetlands. Because most amphibians and many reptiles require aquatic habitats, they are especially vulnerable to alteration or contamination of wetlands (Dodd and Cade 1998, Stebbins and Cohen 1995, Lannoo 1998, Pough *et al.* 1998, Richter and Azous 1995, 2000, Olson and Leonard 1997). Although amphibians have shown some promise as indicators of wetland and/or landscape integrity, no "indices of biotic integrity" based solely on amphibian community composition have yet been developed and validated successfully. Much information on the ecology of tadpoles is summarized by McDiarmid & Altig (1999).

In the decade since our last review, scientific and public concern has increased over the decline of amphibian and reptile populations in North American wetlands (Blaustein and Wake 1995, Cohn 1994, Halliday 1993, Livermore 1992, Wake 1991, Wyman 1990, Phillips 1990, Pechmann *et al.* 1991). Amphibian decline has been well documented (Phillips 1990, Wyman 1990, Wake 1991, Crump *et al.* 1992, Barinaga 1990, Blaustein and Wake 1990), although causes are not yet well understood definitively. A near-absence of long-term studies hinders conclusions regarding the hypothesis that amphibians are experiencing an overall decline due to human activities (Blaustein 1994, Pechmann and Wilbur 1994). Declines have been attirbuted to multiple factors acting singly or in combination (Blaustein and Wake 1990, Sarkar 1996), with diseases and parasites being suggested most often as direct or indirect causes of decline (Carey and Cohen 1999). In the Pacific Northwest alone, amphibian decline or changes in species composition have been linked partly to pathenogenic fungi (Blaustein *et al.* 1994a, Kiesecker & Blaustein 1999), ultraviolet-B radiation (Blaustein *et al.* 1994b, 1995), agricultural runoff (Boyer 1993, Boyer and Grue 1995, Marco et al. 1999), and introduced species (Hayes and Jennings 1986). Investigators also have found some amphibian life stages to be negatively affected by pH and a variety of chemical contaminants (Beattie and Tyler-Jones 1992, Rowe *et al.* 1992, Sadinski and Dunson 1992, Rowe and Dunson 1993, 1995, and see section 7.3 below).

In the past decade a few studies have begun to use amphibian assemblages specifically to indicate the ecological condition of a large series of wetlands, e.g., Richter & Azous (2000). In Minnesota, amphibians were used to represent the condition of landscapes that contained a large wetland/ riparian component (Galatowitsch et al. 1998, Mensing et al. 1998). Sampling 15 wetlands belonging to each of 4 wetland types, the investigators found positive or negative correlations of the following metrics with a site disturbance score and/or various land cover types measured within 500, 1000, and 2500 m of each wetland:

- In small-sized river floodplains: total abundance, abundance of leopard frog
- In medium-sized river floodplains: species richness
- In forest glacial marshes: total abundance, richness
- In prairie glacial marshes: total abundance, richness, abundance of leopard frog

Results are reported by wetland type at: http://www.hort.agri.umn.edu/mnwet/

Other efforts to develop wetland indices of biotic integrity (IBI's) using amphibians are underway in Ohio, Maryland, Maine, and elsewhere.

7.2 Effects of Enrichment/ Eutrophication/ Reduced Dissolved Oxygen

Sublethal responses and mortality of 5 Oregon amphibian species following laboratory applications of nitrate indicated that EPA nitrate criteria for drinking water and/or for protection of warmwater fish are inadequate to protect these amphibians (Marco et al. 1999). In Texas, playa wetlands receiving nutrient-laden feedlot effluent were devoid of amphibians found in natural playas (Chavez et al. 1999). Experiments indicated that effluent had to be diluted to less than 3% strength in order to minimize adverse effects on leopard frog (*Rana pipiens*).

Indirect effects of excessive nutrients can also be important. Shifts in seasonal timing and amount of nutrients that enter a wetland can, over a period of years, increase the relative dominance of algae and/or emergent plants at the expense of submersed plants (see Sections 2 and 3). This in turn can reduce the availability of submersed plants as attachment substrates for amphibian eggs and as cover for larvae (Beebee 1996). It can also diminish dissolved oxygen levels (Tattersall and Boutilier 1999), alter the abundance of aquatic predators, and shift the algal and invertebrate foods available to amphibians (Horne and Dunson 1995b). As a result, species composition and sometimes species richness of amphibian communities can decline as eutrophication becomes severe, but well-designed studies of such effects are few. In Pennsylvania, 10 duckweed-covered wastewater ponds were compared with 10 naturally-occurring ponds over a 19-week period. Egg hatching and survival of all amphibian species was lower in the wastewater ponds (Laposata & Dunson 2000). In the southeastern United States, flatwoods salamanders (*Ambystoma cingulatum*) are not found in wetlands with excessive amounts of algae (Palis 1996).

7.3 Effects of Contaminant Toxicity

Studies of the effects of heavy metals, pesticides, and other toxins on reptile and amphibian communities have mainly been conducted at the species (not community) level of organization. A review of much of the relevant literature was published by Sparling et al. (2000). As a partial starting point for formulating indices of biotic integrity, Schuytema & Nebeker (1996) compiled a database of toxicity information from published literature, for 58 amphibian species as related to 135 chemicals. A similar toxicological database was compiled by Pauli et al. (2000).

Toxicity of aluminum and other metals has been the focus of studies of the embryos and tadpoles of *Bufo americanus* (Birge *et al.* 1992, Freda 1990, Freda and McDonald 1993), *B. canorus* (Bradford *et al.* 1991), *Ambystoma maculatum* (Freda and McDonald 1993), *Hyla crucifer* (Glooschenko *et al.* 1992), *Rana sylvatica* (Freda and McDonald 1993), *R. muscosa* (Bradford *et al.* 1991), and *R. pipiens* (Freda and McDonald 1990, Freda *et al.* 1990, Freda 1991, Freda 1989). Significant variation exists in the suceptability of amphibians to aluminum and pH. Aluminum toxicity was correlated with the mortality of *R. sylvatica* tadpoles but not with mortality of *R. sylvatica*, *Bufo americanus*, and *Ambystoma maculatum* embryos, which were influenced more by pH (Freda and McDonald 1993, Freda et al. 1991). Aluminum treatment at pH 5.0-6.0 resulted in reduced survivorship in *B. canorus* embryos but not *R. muscosa* embryos.

Although not lethal, moderately reduced pH and elevated aluminum was associated with sublethal effects to *B. canorus* and *R. muscosa* embryos, such as reduced growth rates, increased frequencies of developmental abnormalities, and earlier hatching (Bradford *et al.* 1994). Aluminum can ameliorate the toxicity of acidic conditions at some pH levels while becoming toxic at other pH levels. The difference between toxicity and non-toxicity can be quite narrow. Aluminum buffered the toxic effects of acidity on *Rana pipiens* embryos and pre-stage tadpoles in the 4.2- 4.4 pH range but became toxic in the 4.6-4.8 pH range (Freda and McDonald 1990).

The frog, *Hyla crucifer*, was mostly absent in Ontario ponds downwind of a smelting operation, which had higher levels of cadmium, nickel, and sulfate (Glooschenko *et al.* 1992). Cadmium and lead readily accumulated in the frog, *Rana ridibunda* (Vogiatzis and Loumbourdis 1998, 1999) and other amphibian species (Herkovits & Perez 1993, Steele et al. 1999). The fertilization success of *Rana heckscheri* declined markedly when exposed to mercuric chlorides (Punzo 1993a,b). Development was completely blocked at concentrations of 5.0 mg/L of mercuric chloride. Alligators readily concentrate mercury (Heaton et al. 1997, Yanochko et al. 1997) and concentrations tend to be unrelated to the length of sampled individuals, and thus presumably their age (Jagoe *et al.* 1998). Some studies have linked local declines of wetland amphibians with presumed applications of agricultural chemicals in adjoining areas (Berrill *et al.* 1997, Howe *et al.* 1998).

Many synthetic organic compounds affect amphibians and aquatic reptiles. Petroleum derivatives have been noted to stunt tadpole growth of *Hyla cinerea* (Mahaney 1994) as well as reduce development time, growth, and survival in frogs and toads (Pollet et al. 2000). In areas with high oil concentrations, no tadpoles successfully metamorphosed. Northwestern Salamander (*Ambystoma gracile*) egg mortality corresponded with levels of total petroleum hydrocarbons in western Washington (Platin 1994, Platin and Richter 1995). Snapping turtles (*Chelydra serpentina serpentina*) with higher exposures to polychlorinated aromatic hydrocarbons (PAH's) in Ontario and New York had a higher incidence of abnormal development (Bishop et al. 1990, 1995). DNA damage was found in slider turtles (*Trachemys scripta*) and *Chelydra serpentina* that had been exposed to radionuclides and chemical contamination (Lamb et al. 1991, Meyers-Schone *et al.* 1993). Eggs of the turtle *Chelydra serpentina* from sites with the greatest pollution had the highest rates of abnormalities (Bishop *et al.* 1990). The pesticide, esfenvalerate, caused damaging sublethal effects on tadpoles of *Rana pipiens* (Materna *et al.* 1995). Eisler tested the toxicity to frog tadpoles of paraquat (1990a,b), cyanide (1991a), diflubenzuron (1991b), fenvalerate (1992b), zinc (1993), and acrolein (1994). Laboratory exposure of American toad (*Bufo americanus americanus*) and green frog (*Rana clamitans melanota*) to water from a vegetable-growing area in Ontario resulted in a higher rate of tadpole deformities and lower egg hatching rates than exposure to water obtained from upstream of the agricultural area; the water contained high nutrient levels and a mix of pesticides typical of ambient field concentrations (Bishop et al. 1999).

Tests of three forest insecticides (fenitrothion, triclopyr, and hexazinone) on three frog species (*Rana catesbeiana*, *R. clamitans melanota*, *R. pipiens*) in Ontario suggested that none of the species were adversely affected by hexazinone exposure, at least not immediately or observably. Tadpoles of all were sensitive to triclopyr and fenitrothion, with *R. pipiens* being less sensitive than *R. clamitans melanota* and *R. catesbeiana* (Berrill *et al.* 1991). Embryos and larvae of five amphibians (the frogs *Rana sylvatica*, *R. pipiens, R. clamitans melanota*; the toad *Bufo americanus*; the salamander *Ambystoma maculatum*) were exposed to one or both of the pyrethroid pesticides permethrin and fenvalerate, and *Ambystoma maculatum* was found to be particularly sensitive (Berrill *et al.* 1993). Densities of mink frogs (*Rana septentrionalis*) in New Brunswick, Canada, were lower in ponds with greatest exposure to the insecticide fenitrothion (McAlpine et al. 1998). Exposure to sublethal levels of the insecticide carbaryl caused significant and permanent behavioral disruptions in plains leopard frog tadpoles (*Rana blairi*) (Bridges 1997). However, considerable variation was noted in the response of various individuals to this insecticide (Bridges & Semlitsch 2000).

In a study of accumulation of organochlorine chemicals in embryonic turtles, tissue concentrations peaked at or just before hatching and then declined, which is consistent with trends reported in developing sea turtles, fish, and birds (Bishop *et al.* 1995). Morphological abnormalities in Ontario turtles coincided with accumulation of organochlorines in turtle blood (Solla et al. 1998). Deformities in Florida alligators also coincided with organochlorine and PCB contamination, but could not be causally linked to those substances (Guillette et al. 1999). PCB's were found to accumulate to toxic levels in turtles in Mississippi wetlands (Kannan et al. 2000). In Ontario, PCB accumulation also was noted among green frogs (*Rana clamitans*) and snapping turtles (Russell et al. 1997, Bishop et al. 1996). In Illinois, reproductive abnormalities in cricket frogs (*Acris crepitans*) were linked to contamination with PCB's and possibly the herbicide atrazine (Reeder et al. 1998). Atrazine exposure appeared to influence the size and weight of tiger salamanders in North Dakota (Larson et al. 1998) and some reproductive hormones in Florida alligators (Crain et al. 1997).

At times, adult amphibians seem unaffected by ambient concentrations of pesticides and other synthetic organics. In a comparison of frogs in wetlands within apple orchards receiving pesticide (dichlorodiphenyltrichloroethane [DDT]- or endosulfan-related) treatment and those outside of the orchard, no significant effects were observed consistently, despite accumulation of these substances in frog tissue (Harris *et al.* 1998a,b). When the insecticide endosulfan was tested for its toxicity to *Rana sylvatica*, *Bufo americanus,* and *Rana clamitans* embryos and tadpoles in a laboratory setting, the *Rana sylvatica* embryos hatched successfully and displayed no adverse morphological effects during the following 10 days, although nearly all tadpoles had shown paralysis from the insecticide at some point in their development (Berrill *et al.* 1998). In another experiment involving endosulfan, pre-metamorphic mortality of *Bufo americanus* tadpole mortality was high and occurred at the lowest exposure concentrations (0.041-0.053 mg/L) for each species exposed as 2-week-old tadpoles. Newly hatched *Bufo americanus* tadpoles were more tolerant than tadpoles of the other two species, but no clear species differences in sensitivity of 2-week-old tadpoles were apparent. *Bufo americanus* pre-metamorphs exposed to endosulfan did not recover from adverse exposure effects (Berrill *et al.* 1998).

7.4 Effects of Acidification

Excessive acidity damages amphibians both directly (Horne and Dunson 1994b) and as a result of its capacity to mobilize toxic metals and perhaps, by making sodium less available in some soil types (Wyman and Jancola 1991).

Most adult salamander species choose less acidic pools for breeding (Kutka and Bachmann 1990, Whiteman *et al.* 1995). In Ontario, the acid-neutralizing capacity (alkalinity) of 38 wetlands positively influenced the probability of *Rana pipiens*, *R. clamitans*, and *Hyla crucifer* being present (Glooschenko *et al.* 1992). Many researchers have expressed concern that temporary ponds could be the most sensitive freshwater bodies to atmospheric acidification because they are more isolated from inorganic soil buffers, have less dilution, and are directly exposed to acidic rainfall and unbuffered snowmelt (Harvey 1989). In the most acidic spawning ponds on Cape Cod (pH 4.3 to 4.5), spotted salamanders (*Ambystoma maculatum*) suffered complete mortality, which was preceded by gross abnormalities (Portnoy 1990). Similar results were found elsewhere in Massachusetts, in vernal pools (Shortelle *et al.* 1989). Naturally acidic streams in the mountains of North Carolina are virtually devoid of salamander larvae (Kucken *et al.* 1994). When *R. sylvatica*, *Ambystoma maculatum* and *A. jeffersonianum* were exposed to acidic conditions (pH 4.2) in temporary wetlands, *A. jeffersonianum* suffered high mortality for the acute tests and very high mortality for the chronic study. *R. sylvatica* survived the low pH but had reduced growth rates. *A. maculatum* had reduced survival rates in the chronic study (Rowe *et al.* 1992). The trend in acid tolerance seems to be : *R. sylvatica*>*A. maculatum*>*A. jeffersonianum* (Rowe and Dunson 1993, Rowe *et al.* 1992). Embryos of *Ambystoma tigrinum tigrinum* had more than 70% survival at pH 4.5 and above, but suffered much greater mortality at lower pH levels (Whiteman *et al.* 1995). Tables 7.1 and 7.2 provide critical pH values for some common amphibians.

Table 7.1. Lethal and critical pH for embryos of eight amphibian species (adapted from Glooschenko *et al.* 1992).

Species	Lethal pH[1]	Critical pH[2]
Hyla crucifer	3.8-4.2	4.2
Hyla versicolor	3.8	4.3
Rana sylvatica	3.5-4.0	3.9-4.25
Rana pipiens	4.2-4.5	4.6
Rana clamitans	3.7-3.8	4.1
Rana catesbeiana	3.9	4.3
Bufo americanus	3.8-4.2	4.0-4.2
Ambystoma maculatum	4.0-4.5	4.5-5.0

[1] the pH that kills 85% or more of the embryos within a few hours

[2] the pH that causes high embryonic mortality, with effects that are more complex and less direct than lethal effects (Gosner and Black 1957).

TABLE 7.2. Response of several western amphibian embryos to acid water.

The last column is the lowest tested pH without mortality significantly different from controls (adapted from Corn and Vertucci 1992 ,with data from Harte and Hoffman 1989, and Corn *et al.* 1989).

Species	LC50 pH level	No Mortality pH level
Ambystoma tigrinum	5.3	6.1
Bufo boreas	4.5	4.9
Pseudacris triseriata	4.8	5.2
Rana pipiens	4.5	5.0
Rana sylvatica	4.3	4.6

A number of amphibians inhabit peatlands, e.g., *Rana pipiens*, *R. sylvatica*, *Bufo americanus*, *Ambystoma laterale*, *Pseudacris triseriata*, *P. crucifer*, and *Hyla versicolor*. However, acidic water of bogs and marginal fens inhibits embryonic development and hatching of many. Perhaps least affected is *Rana sylvatica,* but even that species is affected in some situations (Karns 1992). The variability in that species' tolerance may depend on genetic or non-genetic maternal factors (Pierce and Wooten 1991).

Concerns have been raised regarding the vulnerability to acidification of western montane wetlands. Acidification makes aluminum and cadmium more mobile and increases their concentration in surface waters. Acidification effects on mobility of lead and inorganic mercury is less, due to the high binding affinities of these metals for humic substances in soils and sediments (Scheuhammer 1991). Amphibians (e.g., Jefferson and spotted salamanders) are known to be sensitive to acidity and elevated concentrations of aluminum found in some acidic ponds (Blancher 1991, Huckabee *et al.* 1989, Ireland 1991, Horne and Dunson 1995b).

However, vernal pools in the mountains of northern Colorado and southern Wyoming rarely had pH levels below 6.0 during the amphibian breeding season (Corn *et al.* 1989). In the California Sierra Nevada mountains, snowmelt water was found to have insufficient acidity to adversely impact *Bufo canorus* and *Rana muscosa* embryos and hatchlings (Bradford *et al.* 1991, 1992). Also, the pulse of snowmelt water probably occurs before breeding begins (Corn and Vertucci 1991). Moreover, no evidence was found to suggest that acidification was affecting Pacific chorus frogs (*Pseudacris regilla*) in the Emerald Lake watershed of the southern Sierra Nevada (Soiseth 1992).

Nonetheless, aluminum released into montane pools as a result of acidification sometimes has harmed embryos, reduced growth rates, and/or caused deformities and premature hatching of native amphibians (Bradford *et al.* 1991, Corn and Vertucci 1991, Lamnicky 1990). Survival of wood frog embryos declined when exposed to aluminum concentrations of 100 µg/L or greater. Boreal toad embryos survived exposure to aluminum concentrations of 400 µg/L (Corn *et al.* 1989). Concentrations of aluminum, sulfate, and zinc were higher in 40 ponds where breeding of Jefferson salamander (*Ambystoma jeffersonianum)* was unsuccessful, whereas alkalinity, copper, dissolved organic carbon, magnesium, sodium, and nitrate were significantly higher in ponds that supported successful breeding. Low pH increased the time until eggs hatched, decreased hatching success, and slowed amphibian development rates. Copper and low pH negatively affected the rate of larval development, whereas aluminum, lead, and sodium did not (Horne and Dunson 1994a). Bullfrog (*Rana catesbeiana*) tadpoles collected from coal ash deposition basins contaminated with As, Cd, Cr, Cu, Se and other elements had reduced number of labial teeth and deformed labial papillae (Rowe et al. 1996). Deformed tadpoles were less able to graze algae, which resulted in lower growth rates.

Amphibian vulnerability to acidification of temporary ponds is largely species-specific (Horne and Dunson 1995a). In temporary ponds of central Pennsylvania, the number of egg masses deposited by *Ambystoma jeffersonianum* was correlated positively with pH and alkalinity and negatively with Al levels. Egg deposition by *A. maculatum* was correlated positively with pH and pond volume and negatively with total cations (Na, K, Mg, Ca, specific conductance) and silicon. Egg deposition by *R. sylvatica* was correlated positively with pond volume and dissolved organic carbon. The results suggest that the breeding success of these amphibians can be predicted in part by abiotic conditions such as pH, aluminum total cations, dissolved organic carbon, silicon, and pond volume (Rowe and Dunson 1993). Mortalities of *Bufo americanus*, *Rana sylvatica*, and *Ambystoma maculatum* embryos were correlated with pH and not to aluminum (Freda and McDonald 1993). The mortality of *Rana sylvatica* tadpoles was correlated both with aluminum and pH. Naturally occurring metals (Al, Cu, Fe, Pb, and Zn) at concentrations analogous to those observed in prior field studies had variable effects on acute exposure and survival for both the Jefferson salamander and wood frog (*Rana sylvatica*) (Horne and Dunson 1995a). Acute exposure to aluminum and copper significantly reduced wood frog survival. Increased water hardness significantly increased acute exposure wood frog survival. Acute exposure mortality of the Jefferson salamander was significantly higher in the aluminum and copper treatments; in toxic metal treatments, survival was higher at the low pH level. Chronic exposure of wood frogs to aluminum and copper at a higher pH level, and lower water hardness level greatly reduced survival. Similarly, chronic exposure of Jefferson salamander larvae to aluminum and copper significantly reduced survival. Newly hatched frog embryos that were exposed to a combination of pH 4.5 and increasing aluminum concentrations experienced increased mortality, reduced body size and swimming speed of tadpoles, and increased predation of tadpoles by dragonfly larvae (Jung and Jagoe 1995). Slower tadpoles are more susceptible to predation by dragonfly larvae (Richards and Bull 1990a).

Excessively high pH, as well as low pH, is detrimental to amphibians. Experiments suggested that the current upper regulatory limit of pH 9 may be inadequate to protect tiger salamanders from detrimental effects of some irrigation and urban wastewater (Abbasi et al. 1989).

7.5 Effects of Salinization

Three studies reported a statistically significant negative correlation between water column conductivity and amphibian species richness (Azous 1991, Platin 1994, Platin and Richter 1995). In general, relatively little is known about thresholds of amphibian tolerance to salinity.

7.6 Effects of Sedimentation/Burial, Turbidity, Shade

A single large pulse of sediments into a northern California stream during construction apparently reduced populations of *Ascaphus truei* (larvae), *Dicamptodon tenebrosus* (paedomorphs and larvae) and *Rhyacotriton variegatus* (adults and larvae) (Welsh and Ollivier 1998). Deposition of silt, especially in combination with motor oil, resulted in reduced growth and earlier metamorphosis of larval mole (*A. opacum*) and tiger (*A. tigrinum tigrinum*) salamanders, as well as increased susceptibility of these species to *Saprolegnia* fungus (Lefcort et al. 1997).

Few studies of sedimentation effects on amphibians have been conducted in wetlands. On one hand, many species require soft sediments as hibernation sites, e.g., painted turtle (*Chrysemys picta*) in sediments 0.50 to 0.95 meters thick in an Ontario pond (Taylor and Nol 1989). On the other hand, excessive sediments (when stirred) impair light penetration of the water column and thus can inhibit growth of algae and especially submersed aquatic plants, which provide cover and attachment sites for amphibian eggs.

7.7 Effects of Vegetation Removal

Gross classifications of vegetation form (e.g., Cowardin class) are only moderately useful for describing habitat of some amphibians (Munger *et al.* 1998). In oligotrophic lakes of Quebec, vegetation structure had a low capacity to predict the occurrence of bullfrogs (*Rana catesbeiana*), mink frog (*R. septentrionalis*), and green frog (*R. clamitans melanota*) (Courtois *et al.* 1995), but was important to amphibians using a wetland constructed to treat mining wastes (Lacki et al. 1992). In the Puget Sound Basin of Washington, surveys of 19 wetlands found no statistically significant correlations between amphibian richness and vegetation form (Richter & Azous 1995). Aquatic amphibians appear to require particular types of submersed plants as deposition sites for their eggs. Plant stem diameter (e.g., stems <3 mm preferred by northwestern salamander)(Richter and Roughgarden 1995) is apparently more important than plant species (Richter 1997). Density of submerged plants also is important. A survey of 40 wetlands in the Puget Sound found more native species among wetlands containing dense emergent vegetation (Adams and Bury 1998). Dense vegetation may help protect larval young of native aquatic amphibians from larger predators.

Vegetation condition in surrounding buffer zones and watersheds can be at least as important to amphibians as vegetation condition within wetlands. Amphibians were less common in Pennsylvania stream corridors where vegetation had been removed from surrounding areas (Croonquist 1990, Brooks & Croonquist 1990). After a clearcut of a bottomland forest wetland in Louisiana, mole salamanders (*Ambystoma talpoideum*) in the vicinity had lower survival rates (Raymond and Hardy 1991). Clearcutting in a southern Alabama bottomland hardwood wetland resulted in only brief depression of species richness, but salamander diversity and abundance were greatly reduced whereas frog and toad species increased (Clawson et al. 1997). In a bottomland forest wetland in South Carolina, salamanders were much more common in mature stands than in clearcut areas (Phelps & Lancia 1995). Uncut sites also had more gray treefrogs (*Hyla chrysoscelis*), bronze frogs (*Rana clamitans*), and box turtles (*Terrapene carolina*). Among 16 ephemeral ponds situated within tree plantations in New Brunswick, Canada, higher densities and rates of recruitment of several amphibian species were found in the ponds situated closer to natural forest (Waldick et al. 1999). The juxtapositioning of ponds and forest also was found to be important to amphibian diversity in Shenandoah Mountains of Virginia (Mitchell et al. 1997). Salamander populations in the Appalachians may require 50-70 years to recover from clear-cutting (Petranka *et al.* 1993). Recolonization of wetlands denuded by the Mount St. Helens volcano in Washington is projected to take up to 100 years due to slow dispersal rates of amphibians and the distance from source habitats (Hawkins and Sedell 1990).

In Maine forests, the abundance of frogs and salamanders (*Rana sylvatica* and *Ambystoma maculatum*) declined along a gradient from mature forest-interior habitat (70-90 years old) to recently clearcut habitat (2-11 years old) (deMaynadier and Hunter 1999). In one study in northern California, streams flowing through uncut forests contained all 4 common amphibians whereas only one of the streams flowing through cut forests contained all 4 amphibians. Only 2 uncut sites had fewer than 3 amphibians whereas 11 of the logged sites had no amphibians (Welsh 1990). In Oregon, amphibian species richness was higher in streams flowing through undisturbed forests than in streams flowing through logged forests (Corn and Bury 1989). Undeveloped vegetated buffers of 30-95 meters have been suggested to help maintain diversity of amphibians (Rudolph and Dickson 1990) and are at least equally important to turtles (Burke & Gibbons 1995).

Effects on amphibians and reptiles of removing surrounding wetland vegetation are not always negative. The affinity of many reptiles for warm microclimates led to an overall increase in herptefauna diversity in clearcut plots in a South Carolina bottomland wetland landscape (Phelps and Lancia 1993, Perison et al. 1997). Among 37 Michigan wetlands studied over 20 years, two-thirds of the local species extinctions occurred in wetlands where forests had grown up in the surrounding area during that period, perhaps shortening the annual duration of inundation of understory vernal pools and cooling the substrate (Skelly et al. 1999).

7.8 Effects of Thermal Alteration

Water and air temperatures can have different effects on larval vs. adult amphibians. Excessively warm temperatures can dessicate amphibians once pools dry up (Shoemaker *et al.* 1992).   In winter, painted turtles (*Chrysemys picta*) sometimes move about so they can maintain a temperature of 4-6°C, but must restrict such movements if ice cover causes anoxia in pond sediments (Taylor and Nol 1989). Some aquatic salamanders in the Pacific Northwest appear to choose northern shores of ponds and wetlands for egg-laying, presumably due to greater solar exposure and warmer microclimate (Richter 1997). Northern water snake (*Nerodia sipedon*) is a wetland-dependent species that may be especially sensitive to temperature (Robertson and Weatherhead 1992).

7.9 Effects of Dehydration/Inundation

Most amphibians require moist conditions and cannot tolerate prolonged dry periods. Amphibians also can be extremely specific in their water depth requirements, especially for oviposition (Miaud 1995). Desiccation of seasonal pools, especially when it occurs ahead of normal seasonal schedules, can ruin breeding success of amphibians (Rowe and Dunson 1993). This is partly because many amphibian species disperse only short distances (Berven and Grudzien 1990). Many amphibian species survive long-term droughts or floods by maintaining populations scattered across a variety of wetlands of different depth and water permanence.

The availability of numerous scattered wetlands can serve as a “cushion” against effects of localized drought. Indeed, some frog and toad species living in relatively intact landscapes seem mostly unaffected at a population level by significant periods of drought (Dodd 1995). In contrast, when wetland alteration increases the distances between the remaining fishless wetlands most suitable for amphibians, and when terrestrial vegetation along dispersal routes is replaced by roads or other unsuitable habitats, amphibian populations recover slowly or not at all from droughts they might otherwise survive (Pounds and Crump 1994). Some amphibians in the Florida Everglades survive the dry season in limestone solution cavities beneath the land surface, and may live up to several months in these waters. Increased extraction of this water for human use has made many of these potential dry-season refuges unavailable to aquatic animals (Loftus *et al.* 1992).

Both prolonged desiccation and extreme floods can also increase opportunities for invasion of wetlands by exotic plant species. Consequently, patterns of vegetation typically become more homogeneous, and suitability of amphibian habitat as well as prey abundance may decline (Munn and Brusven 1991, Ludwa 1994). Currents associated with floods can hinder breeding and egg deposition for many amphibian species that require still water (Lind *et al.* 1996, Richter 1997), but also distribute woody debris and coarse sediments that are important components of amphibian and reptile habitat. Naturally-occurring floods in larger rivers create a geomorphically diverse mosaic of floodplain wetlands that cumulatively can support a similarly diverse array of amphibian and reptile species (Galat et al. 1998).

A virtual absence of water level variation also can be indirectly detrimental to some amphibians, because without occasional drying of substrates, nutrient cycles in some wetlands stagnate, vegetation patterns become more homogeneous, and suitability of habitat may decline.

Effects on Species Richness

In South Carolina wetlands, amphibian species richness was statistically correlated with seasonal permanence (Snodgrass *et al.* 2000). In Indiana, a survey of 30 forested wetlands found the greatest amphibian species richness in wetlands of intermediate permanency (Kolozsvary and Swihart 1999), although another Indiana study found amphibian richness to be greatest when wetlands were located near permanent water bodies (Brodman and Kilmurry 1998). The number of amphibian species in wetlands of the Puget Sound Basin of Washington was related more to water level fluctuations than to vegetation form, with lowest richness occurring when springtime fluctuations exceeded 20 cm (Richter 1997). Amphibian richness among 12 temporary wetlands in Florida was related more to the presence of fish (*Lepomis* sp.) than to water source (Babbitt and Tanner 2000). Amphibian breeding success appeared to depend largely on the timing of inundation, not as much on its duration. Wetlands connected to fish bearing waters had fewer amphibian species, a phenomenon that has been noted elsewhere and is attributable to significant fish predation on amphibian eggs and larvae (Hecnar & M'Closkey 1997, 1998).

Effects on Species Composition

Although some amphibian species are adapted to short (less than 3 months) inundation of wetlands, many require longer permanence to produce adequate offspring (Pfingsten and Downs 1989, Tyning 1990, Conant and Collins 1991). For example, in Michigan, the distribution of 3 of 14 species were significantly affected by pond permanence (Skelly and Meir 1997). Among depressional wetlands of the upper Atlantic Coastal Plain, sirens (*Siren intermedia* and *S. lacertina*) and amphiumas (*Amphiuma means*) were mostly found in wetlands with longer durations of inundation. Biological interactions between these species also was suspected of limiting their geographic distribution (Snodgrass *et al.* 1999).

In northwestern Nevada, severe drought and accompanying high temperatures killed many Pacific chorus frogs (*Pseudocris regallis*) (Weitzel and Panik 1993). During a 2-year drought in the state of Washington, a local population of painted turtle (*Chrysemys picta belli*) suffered a 70% decline (Lindenman and Rabe 1990). This appeared to be due to both mortality and emigration from the wetland. Growth also was suppressed, but recovered following improved conditions. The average sizes of cohorts was not affected. However, in temporary forest pools in Mississippi, only 47% of the amphibian cohorts inhabiting the pools appeared metamorphosed before the pools dried out (Bonner et al. 1997). In central Pennsylvania, spotted salamanders (*Ambystoma maculatum*) -- which are among the last species to breed in the spring -- were noted as being especially sensitive to drought (Rowe and Dunson 1993). In a Florida population of striped newt (*Notophthalmus perstriatus*) larger individuals became proportionally more dominant after a severe drought (Dodd 1993b). Also in Florida, natural hydroperiod alterations did not influence the activity of swamp snakes (*Seminatrix pygaea*) in wetlands. Drought had little direct effect on overland migration or body condition, and only caused snakes to leave or to shorten the amount of time they spent within a wetland (Dodd 1993a). Water level drawdowns conducted in the autumn for wetland management, flood control, or other reasons can cause high mortality among juvenile overwintering turtles (due to freezing) if the drawdowns follow abnormally high late-summer water levels that attracted turtles (Galat et al. 1998).

Relative dominance in a wetland of amphibian species that metamorphose quickly into a terrestrial stage may indicate short duration of inundation. In contrast, relative dominance of slow-growing species can indicate significant co-occurrence of predatory fish and invertebrates, which are typically associated with long duration of inundation (Wellborn *et al.* 1996, Schneider and Frost 1996, Schneider 1997). Some amphibians seem especially susceptible to predation as they move from drying fringes to deeper waters of the same wetland, which often tend to have less protective vegetation. For example, Pacific tree frog (*Pseudacris regilla*), red-legged frog (*Rana aurora*), spotted frog (*R. pretiosa*), northern leopard frog (*R. pipiens*), and western pond turtle (*Clemmys marmorata*) appear to be more susceptible to predation by bullfrog (*R. catesbeiana*) as they move to more open water (Leonard *et al.* 1993, Hallond *et al.* 1995).

Extended floods, by inundating shoreline turtle nests or alligator nest mounds, can diminish or ruin a season's recruitment of young (Kushlan and Jacobsen 1990, Tucker et al. 1997). In Oregon, numbers of a mostly terrestrial salamander (ensatina, *Ensatina eschscholtzii*) were less in frequently-flooded riparian habitats of red alder (McComb *et al.* 1993a, McComb *et al.* 1993b), second growth conifer (Gomez and Anthony 1996), and unmanaged Douglas fir (Aubry and Hall 1991, Gilbert and Allwine 1991).

7.10 Effects of Habitat Fragmentation and Other Stressors

Amphibian and turtle diversity, measured at a local or regional scale, is often severely affected by filling of -- or intentional connecting of (with ditches) -- small, somewhat isolated, temporarily or seasonally inundated wetlands. Such wetlands are often the first to be ignored by state and federal regulatory programs, or altered in exchange for protection/restoration of larger wetlands as part of mitigation banking agreements. Large wetlands do not necessarily support a wider variety of amphibians, as indicated by surveys of 97 Ontario ponds (Hecnar and M'Closkey 1996), 19 Washington wetlands (Richter and Azous 1995), and 22 South Carolina wetlands (Snodgrass *et al.* 2000).

An analysis of wetland spatial data in Maine, using demographic models for amphibians and turtles, indicated that cumulative loss of many small wetlands, by increasing the distance between wetlands and reducing dispersal success of several species, would result in eventual extinction of many species, especially turtle species (Gibbs 1993). A similar simulation based on landscapes in the Atlantic coastal plain concluded that isolated wetlands as small as (0.2 hectare) need to be protected, and spatial patterns of wetland distribution taken into account, if regional biodiversity of amphibians is to be maintained (Semlitsch & Bodie 1998). Connecting isolated wetlands with ditches often introduces predatory fish or carnivorous amphibians to somewhat isolated amphibian populations, with consequent shifts in species composition and sometimes a general decline in abundance of native species (Babbitt and Tanner 2000).

Loss of vegetated uplands that connect isolated wetlands can hinder an amphibian population's ability to recover from drought (Pounds and Crump 1994), disease (Bradford 1991) low reproductive rates (Sinsch 1992), and other wetland alterations (Dodd and Cade 1998). Probability of occurrence of several amphibian species in Indiana was strongly associated with proximity to wetlands (Kolozsvary & Swihart 1999). In another Indiana study, amphibian richness was greater in wetlands located close to forested areas (Brodman and Kilmurry 1998). A similar result was found in Washington (Richter & Azous 2000). In New Mexico, 3 wetland-associated toad species fed extensively on carabid beetles in surrounding terrestrial habitats (Anderson *et al.* 1999). A lack of suitable upland habitat adjoining some southern Illinois wetlands was partly blamed for absence of some species (Burbrink et al. 1998). The width of the riparian corridor connecting the wetlands appeared to be less important to supporting a diverse amphibian community than local habitat heterogeneity and distances between source wetlands.

Changes in land cover leading to increased isolation of wetland breeding habitats from each other, or from essential upland habitats, have been suggested as a cause of decline of many amphibian species (Blaustein *et al.* 1994c, Sjögren-Gulve and Ray 1996), including mountain yellow-legged frog (*Rana muscosa*) (Bradford, Tabatabai, and Graber 1993), Oregon spotted frog (*Rana pretiosa*) (Orchard 1992, Azous & Richter 1995, McAllister and Leonard 1990, 1997) and western pond turtles (*Clemmys marmorata*) (Gray 1995). Among 21 Minnesota glacial marshes, amphibian richness was lower in marshes that were more isolated (Lehtinen *et al.* 1999). Among 37 Michigan wetlands studied over 20 years, the number of amphibian species extinctions and colonizations was greater among the more isolated wetlands (Skelly et al. 1999). However, among 19 Puget Sound Basin wetlands of Washington, amphibian richness was not necessarily greater in wetlands that were less isolated from (closer to) other water bodies (Richter & Azous 2000).

Even when they are connected by water, wetlands might be considered "isolated" when the distances between them exceed the typical dispersal distance of amphibian and turtle species they otherwise are capable of supporting. Adults of 6 salamander species used habitats an average of 125 meters from the ponded edge of a wetland in Missouri (Semlitsch 1998). Dispersal distance for many wetland amphibians is <0.3 km (Gibbs 1993; Semlitsch 1998, Semlitsch and Bodie 1998). Semlitsch (1998) recommended that natural land cover be protected within an average distance of 164 m from the edge of wetlands to protect 95% of the ambystomid salamander populations. Non-ambystomid salamanders (newts), frogs, and toads may have greater requirements because of typically greater dispersal distances. Even when habitat or water quality within or around a wetland diminishes, amphibians may still attempt to breed there due to strong behavioral attachment to natal sites (Karns 1992).

Although more protective than developed land uses, it is uncertain whether vegetated upland buffers and/or upland corridors that link wetlands are sufficient to protect some amphibians and turtles from becoming locally extinct. Impacts to amphibians occur because developed land is characterized not only by less habitat space and greater wetland isolation, but also by alteration of wetland water quality/hydrology and an increased number of edges between forest and openland, which some species avoid (Gibbs 1998). Among 21 Minnesota glacial marshes, amphibian richness declined as the proportion of urban land cover increased at all spatial scales (Lehtinen *et al.* 1999). Similar results were found in surveys of frogs or amphibians generally in wetlands of the Seattle area (Azous and Richter 1995, Richter and Azous 1995), Connecticut (Gibbs 1998), and Wisconsin and Iowa (Knutson *et al.* 1999). Specifically, land cover alteration is believed to have negatively impacted spotted frogs (*Rana pretiosa*) in Washington (McAllister and Leonard 1991) and red-legged frogs (*R. aurora draytonii*) in California (Hayes and Jennings 1988). Even outside of urban areas, other areas of relatively homogeneous land cover, such as grazed rangeland and agricultural land, have been shown by some studies to support lower amphibian abundance and/or richness (Mensing *et al.* 1998, Hecnar 1997, Bonin *et al.* 1997, Delis *et al.* 1996, Bishop et al. 1999).

However, land cover alteration does not inevitably diminish richness at a local scale. In south-central Florida, an area that was comprised of ditches, pastures, ponds, and orange groves contained as many native reptile and amphibian species as more natural reference areas (Meshaka 1997). In playa wetlands of Texas, frog occurrence did not appear to correlate with land cover or the presence of irrigation pits (Anderson *et al.* 1999). In Wisconsin, amphibian abundance was associated positively with both agricultural lands and with forested lands (Knutson et al. 1999). Thus, the configuration of suitable habitat, which reflects the ability of individuals to safely disperse, may be equally or more important than total habitat area. Effects also may depend on the particular species, with some being more dependent on wetlands and other natural environments.

Many of the impacts found in developed landscapes may be attributed to roads (Langton 1989, Fahrig *et al.* 1995, Gibbs 1998). Vehicular traffic can lead to amphibian declines either through direct mortality and increased exposure to predators (Ashley and Robinson 1996), or as amphibians avoid crossing roads, thus reducing gene flow among populations as evidenced by greater genetic distancing (Reh 1989). Amphibian species richness in 21 Minnesota glacial marshes was less at all spatial scales having greater road density, in 2 ecoregions (Lehtinen *et al.* 1999). In Massachusetts, suburban highways affected amphibian populations more than 100 m from the road (Forman et al. 2000).

Constructed and restored wetlands have been surveyed for amphibians only occasionally. In one instance, a constructed wetland being used to treat acid-mine drainage also supported more amphibian species than nearby natural wetlands, and also had greater abundance of green frogs (*Rana clamitans*) and pickerel frogs (*Rana palustris*) as well as a variety of snakes attracted by the large frog prey base and the availability of den sites amid mining rock debris. Apparently the acidity did not reach toxic levels.

Several studies documented the avoidance of some breeding ponds that contained potential predators. American toads avoided laying eggs in ponds that contained wood frog tadpoles, which feed on the eggs and larvae of toads (Petranka *et al.* 1994). Wood frogs (*Rana sylvatica*), spotted salamanders (*Ambystoma maculatum*), and Jefferson salamanders (*A. jeffersonianum*) seem to be almost entirely dependent on fish-free wetlands for breeding (Rowe and Dunson 1993).

Introduced reptiles and amphibians have caused the decline of some native amphibians. The bullfrog (*Rana catesbeiana*), a large anuran that regularly consumes smaller frogs, was introduced to parts of the western U.S. decades ago and is suspected of contributing to the decline of northern leopard frogs (*R. pipiens*) and red-legged frogs (*R. aurora*) (Panik and Barrett 1994, Lawler et al. 1999). However, other evidence suggests the distribution of native amphibians among Washington wetlands is not linked tightly to bullfrog presence (Adams et al. 1998, Adams 1999), and predatory fish can be at least as important as predators. Introduced fish eliminated populations of mountain yellow-legged frog (*Rana muscosa*) a century ago in many lakes and streams (Bradford et al. 1991, Graber 1993, Knapp & Matthews 2000). However, among 21 Minnesota glacial marshes, species richness was greater in wetlands that contained fish and tiger salamander (*Ambystoma tigrinum*) (Lehtinen *et al.* 1999).

Native turtles possibly are being affected by introduced turtles that compete for habitat and food, as well as spread diseases and parasites (Bury 1994). Decline of western toad (*Bufo boreas*) has been hypothesized to be the result of increased incidence of the pathogenic fungus, *Saprolegnia* Blaustein *et al.* (1994a). Another hypothesis is that the decline of both western toad and Cascade frog (*Rana cascadia*) is attributable to ultraviolet-B radiation (Blaustein *et al.* 1994b). Field experiments revealed that fungus reduced the hatching success of three Oregon anurans most dramatically when the eggs were also exposed to elevated levels of ultraviolet-B radiation (Kiesecker & Blaustein 1995).

Human recreation negatively affected populations of North American wood turtles (*Clemmys insculpta*) in part of Connecticut (Garber and Burger 1995). Wood turtle populations declined where habitat was opened to recreation (i.e., hiking and fishing), perhaps partly because of illegal collection of individuals. Populations remained stable in habitats where recreation was restricted. Legal harvest of 50% of the annual production of alligators in 2 Florida lakes did not significantly affect distribution of age classes within the alligator population (Rice et al. 1999).

7.11 Wetland Monitoring

Spatial and Temporal Variability

Choice of appropriate sample sizes depends on measured variation in the target taxa and metrics. Such coefficients of variation are summarized from various amphibian studies elsewhere at: http://www.mp2-pwrc.usgs.gov/ampCV/ampdb.cfm

At a landscape scale in natural landscapes, the distribution of many amphibians appears to be influenced mainly by suitability of adjoining habitats, water persistence, predator presence, and their interaction (Wellborn *et al.* 1996, Skelly and Meir 1997, Pfingsten and Downs 1989, Tyning 1990, Conant and Collins 1991). Wetlands with greater persistence of water tend to support more animals that prey on amphibians, and often have fewer amphibian species.

Temporally, the populations of most amphibian and reptile species in natural landscapes fluctuate greatly from year to year, sometimes with no obvious cause (Pechmann *et al.* 1991, Dodd 1992, Stone et al. 1993, Cohn 1994). However, substantial turnover may occur in the species composition of individual wetlands, even in the absence of strong human influence. For example, in a study of 14 amphibian species in 37 Michigan wetlands, 2 surveys conducted about 20 years apart recorded 40 colonizations and 34 extinctions, with little overall net change in the breeding populations of most species (Skelly et al. 1999). This natural temporal variability needs to be accounted for when attempting to interpret trends from monitoring data.

Techniques and Equipment

Equipment and methods used to sample amphibians and/or turtles in wetlands include, but are not limited to:

- call surveys (see: http://www.im.nbs.gov/amphib/naampapp1.html)
- timed searches (Croonquist and Brooks 1990, Petranka *et al.* 1993)
- dip net sweeps (Anderson *et al.* (1999)
- drift fences (Dodd 1991, Brenner et al. 1992, Buhlmann.1999)
- pitfall traps (McComb et al. 1991, Mitchell et al. 1993)
- box traps (Babbit and Tanner 2000)
- funnel traps (Richter 1995)
- minnow traps (Kolozsvary and Swihart (1999)
- shelter boards (Grant et al. 1992)

Methods for designing amphibian studies, as well as sampling methods, equipment, and data interpretation, are described by Heyer *et al.* (1994), Fellers & Freel (1995), and Olson et al. (1997). Much of this information was also summarized in Adamus and Brandt (1990), by EPA's amphibian assessment group (http://www.epa.gov/owow/wetlands/bawwg), and by the USGS's NAAMP program: http://www.mp1-pwrc.usgs.gov/amphibs.html

Metrics for Assessing Impacts to Amphibian Communities

Few published studies have examined specific metrics applicable to using amphibians and reptiles for monitoring wetland condition. Because most individual wetlands have few amphibian and turtle species, richness and species composition are best employed as metrics of ecological condition at landscape scales, e.g., to wetland "complexes," rather than to individual wetlands. Nonetheless, within individual wetlands, rates of deformities and total abundance of tadpoles, turtles, and aquatic salamanders can often be used as indicators of wetland condition, provided natural reference conditions have first been adequately measured.

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Section 8: Birds

8.1 Use As Indicators

This section addresses birds that are closely associated with inland wetlands and riparian areas. This includes waterfowl, wading birds, shorebirds, and many songbirds. For a general discussion of the topic based on pre-1990 scientific information, and for discussion of advantages and disadvantages of using birds as indicators of wetland integrity, readers should refer to Adamus and Brandt (1990). A recent book by Weller (1999) provides a good overview of wetland bird ecology and behavior.

In contrast to most other flora and fauna, there have been no recent publications demonstrating use of observation-based bird IBIs (indices of biotic integrity) specifically in wetland or riparian settings. However, several studies have contrasted bird communities in urban/developed settings with similar habitats in rural/undeveloped settings, both at individual paired sites (e.g., Craig & Barclay 1992, Dowd 1992) and at a regional scale that encompasses gradients of human influence (e.g., Croonquist & Brooks 199, Blair 1996, Flather & Sauer 1996, Miller et al. 1997, Galatowitsch et al. 1998, O'Connell et al. 1998, 2000, Whited et al. 2000, Cam et al. 2000). As summarized by Adamus (2001), these studies have clearly supported the utility of employing bird species composition -- and wetland birds in particular -- as an indicator of land cover alteration, habitat fragmentation, and other human influences at multiple scales.

8.2 Effects of Enrichment/ Eutrophication/ Reduced Dissolved Oxygen

Eutrophication can indirectly affect wetland bird community composition by altering the vegetation structure and availability of prey items. For example, fish production is generally greater in Canadian lakes that have at least moderate nutrient levels, and distribution of fish among lakes largely determines local distributions of Common Loon (Kerekes 1990). However, many waterfowl that feed on aquatic invertebrates avoid lakes with fish, because fish can decimate populations of invertebrates most important to ducks (McNicol and Wayland 1992). Moderately elevated nutrient levels also spur the growth of submersed macrophytes important as food for ducks, as well as supporting more aquatic insects that are especially important as food for ducklings and for aerial foragers like swallows. However, excessive nutrients cause algal blooms that can kill fish, decimate macrophytes by blocking light, and reduce visibility to birds of food items located under the water surface. Such a situation has been documented in the Chesapeake Bay (Perry & Deller 1996). Excessive nitrates have been implicated in deaths of some frogs, which are significant prey for many wetland birds (see Amphibians chapter). Northern Shoveler and Eared Grebe were positively associated with phosphorus in a survey of wetlands in interior British Columbia (Savard et al. 1994).

Waterbird abundance and biomass were positively correlated in 46 Florida lakes with levels of phosphorus, nitrogen, and chlorophyll. There also was a positive correlation of waterbird richness with phosphorus, after accounting for nutrients contributed to the lakes by the birds themselves (Hoyer and Canfield 1994). Wetlands constructed for wastewater treatment are often heavily used by waterbirds (Frederick and McGhee 1994). Surveys of 92 British Columbia ponds reported that densities of most of the 17 breeding duck and grebe species were associated positively with total dissolved nitrogen. Total dabbling duck density was correlated positively with total dissolved nitrogen (Savard et al. 1994). Surveys of 837 river corridors in England and

Scotland identified 23 (of 29) waterbird species whose breeding abundance was associated with unspecified water quality variables (Rushton et al. 1994). British surveys of 109 estuaries also identified trophic status as a factor possibly influencing species composition of shorebird assemblages (Hill et al. 1993). A survey of 95 inland wetlands in Australia identified 15 (of 61) waterbird species whose occurrence was somewhat related (negatively) to higher phosphorus levels (Halse 1993).

The parasitic nematode, *Eustrongylides ignotus*, which has only been found in disturbed and enriched wetlands (Spaulding and Forester 1993), negatively affects the health of adult wading birds and the survival of nestlings (Spaulding et al. 1993).

8.3 Effects of Contaminant Toxicity

Several new studies examined effects of contaminants on individual bird species, but seldom on entire bird communities within wetlands.

Several instances have been documented of wetland birds being directly poisoned by insecticides applied at recommended rates (e.g., Flickinger et al. 1991 -- parathion). In North America, recent attention has focused on effects on birds of fenthion, diazinon, phorate, chlorfenapyr, and chlorpyrifos. Many pesticides are more likely to affect birds by altering their habitat and foods than by being directly toxic. Insecticide-related reductions in invertebrate foods of waterbirds have been documented in prairie wetlands (Tome et al. 1990, 1991, McCarthy and Henry 1993, Martin and Solomon 1990). In contrast, application of the non-chemical insecticide *Bacillus thuringiensis* subsp. *israelensis* (Bti, applied as Vectobac-G granules) and methoprene had no detectable effect on breeding birds in Minnesota (Hanowski et al. 1997). Herbicides have been applied to wetlands to alter vegetation structure and species composition, with consequent shifts in bird species composition (Solberg & Higgins 1993, Linz et al. 1997). Information on pesticides in prairie wetlands was compiled by Facemire (1992).

Much research has continued to focus on the effects of selenium on waterfowl in western states (e.g., Hoffman et al. 1996). Biogeochemical conditions favoring the release of selenium into wetlands are found throughout the arid regions of the western states and threaten bird communities in many wetlands that are part of the Pacific and Central Flyways (Lemly *et al.* 1993, Stephens *et al.* 1992, Paveglio *et al.* 1992). Agricultural drainage, irrigation, and natural waters can leach selenium from many western soils. Subsurface irrigation is the most widespread and biologically important source of selenium toxicity for waterfowl, including the waterfowl in six national refuges (Lemly *et al.* 1993, Naftz *et al.* 1993, Ohlendorf *et al.* 1990, Feltz *et al.* 1991). However, Barn Swallows that aerially forage for insects over contaminated areas apparently suffered no adverse reproductive effects (King et al. 1994). Selenium is often accompanied by boron, which also is toxic to ducklings (Stanley et al. 1996).

Contamination of fish-eating terns by selenium and mercury was documented by one study that found significantly higher levels of these metals in breast feathers of birds in North Temperate breeding areas than in feathers of birds just returning from South Temperate wintering grounds (Burger *et al.* 1992). Concentrations of mercury in feathers of 92 southern Florida wading birds indicated that body concentrations were probably high enough to interfere with reproductive success (Sundlof et al. 1994, Beyer et al. 1997). Mercury concentrations vary with geographic locations but also with diet and age of the bird (Sundlof et al. 1994, Beyer et al. 1997). Species that eat larger fish, and older birds, tend to have the highest Hg concentrations.

Breeding songbird richness near zinc- and copper-contaminated Montana wetlands was not significantly less than in uncontaminated reference areas (Linder et al. 1994, Pascoe et al. 1994), but reproductive success of most species was not measured. Another metal -- lead (Pb) -- has been documented to accumulate in Tundra Swans feeding from wetland sediments contaminated by mining and smelting waste (Beyer et al. 1998), reaching levels capable of killing birds (Blus *et al.* 1991). Kendall *et al.* (1996) review the effects of Pb on raptors and upland, nongame birds. Winter survival rates for immature Canvasbacks exposed to Pb in Louisiana were lower than those not exposed to Pb. In all, 16% of the immature canvasbacks that were examined had died from Pb they absorbed by ingesting lead shot in the water (Hohman *et al.* 1995). Other evidence of bird toxicity from lead shot was reported by Havera *et al.* (1992), Hohman *et al.* (1993), and Peters and Afton (1993).

Oil spills can, of course, severely impact wetland bird populations (e.g., Wiens et al. 1996, Burger 1997). The effects of synthetic organic compounds on birds also are receiving increased attention. Past exposure to organochlorine compounds, PCB, dioxin, and other contaminants was documented in Black-crowned Night Herons by measurement of cytochrome P450 (Rattner et al. 1994, 1996, 1997). Using the same biomarker, chronic exposure to petroleum hydrocarbons in oil field brines was documented in Western Sandpipers in Texas, but no acute effects were apparent (Rattner et al. 1995). Planar halogenated hydrocarbons (PHH) have caused embryotoxicity, congenital deformities, and poor hatching success in Forster's Terns (Tillitt *et al.* 1993). PCBs, dibenzodioxins, and polychlorinated dibenzofurans have been implicated as contributors to the continued decline of Common and Forster's Terns in the Great Lakes, whereas some populations of Double-crested Cormorants, Herring Gulls, and other colonial, fish-eating bird populations appear to be increasing as contamination of their particular food chains diminishes (Giesly *et al.* 1994). Detrimental reproductive effects have been documented of organochlorines on Herring Gulls (Ewins *et al.* 1992), dioxins on Great Blue Herons (Hart *et al.* 1991), dioxins and furans on Wood Ducks (White and Seginack 1994, 1995), PCB's in American Kestrels, and petroleum in Mallards (Holmes and Cavannaugh 1990). Advantages and disadvantages of using particular biomarkers of chemical exposure in birds are discussed by Fox (1993).

8.4 Effects of Acidification

Acidification of wetlands affects birds primarily because it reduces calcium availability (important for egg development), potentially increases toxic metal availability, and alters the species composition and occasionally the abundance of aquatic insects, submersed plants, and fish that are important foods for waterfowl.

Reduced availability of calcium-rich foods and the change in types of available food can diminish egg shell thickness and generally reduce the reproductive success of waterbirds in wetlands (Sparling 1990, 1991, Blancher and McNicol 1991, St. Louis *et al.* 1990, Albers and Camardese 1993) and in streams (e.g., dippers in England: Tyler and Ormerod 1992, Ormerod and Tyler 1991). However, when acidification removes fish from wetlands the abundance of insect prey can increase (McNicol et al. 1990, Blancher and McNicol 1991, Blancher et al. 1992) as can selection of the most productive wetlands by hens with young broods (Parker *et al.* 1992, Merendino *et al.* 1992, Merendino & Dennis 1993, Merendino and Ankney 1994), and the number of waterfowl broods (McNicol and Wayland 1992). Nonetheless, insect densities do not necessarily increase the reproductive success of aerial-foraging insectivorous species, e.g., Tree Swallows, which are potential vulnerable to chemical bioaccumulation (St. Louis *et al.* 1990, St. Louis and Barlow 1993, Blancher and McNicol 1991, Froese et al. 1998). Overall, calcium deficiency appears to affect birds in acidified wetlands to a greater degree than metal toxicity (Albers and Camardese 1993). Breeding pairs of 15 waterfowl species are more abundant in Ontario wetlands with > 40 ppm total alkalinity than in less alkaline wetlands (Dennis *et al.* 1989, Merendino *et al.* 1992). In British Columbia as well, densities of several breeding duck species were greater in ponds with higher levels of conductivity and calcium (Savard et al. 1994).

8.5 Salinization

Highly saline or alkali conditions are detrimental to some invertebrate and plant foods used by many duck species, and high salinity is directly toxic to -- or impairs the growth of -- young ducklings (Clark & Nudds 1991, Moorman *et al.* 1991). Those sensitive waterbirds may visit saline wetlands, but often only when fresher wetlands are available nearby (Lokemoen and Woodward 1992, Woodin 1994, Adair *et al.* 1996). Breeding densities of most duck and grebe species in interior British Columbia were greater in ponds with higher conductivity, but marsh-nesting species were unaffected (Savard et al. 1994). In a survey of part of the Canadian prairie, 2% of the wetlands were found to be potentially too saline to support waterfowl reproduction (Leighton and Wobeser 1994).

Nonetheless, a few waterbird species occur regularly at very high densities in alkali wetlands during the breeding season and/or migration, e.g., American Avocet, Black Stilt, Snowy Plover, phalaropes, Killdeer, Horned Grebe, Tundra Swan; White-rumped, Semipalmated, and Baird's sandpipers (Eldridge and Krapu 1993, Earnst 1994, Jehl 1994, Kingsford and Porter 1994, Savard et al. 1994, Oring & Reed 1997, Rubega & Johnson 1997, Warnock 1997). These relatively salt-tolerant species also occur in less saline wetlands, but their abundance often is greatest in hypersaline wetlands, and is related to sharp seasonal peaks in the abundance of brine shrimp and other salt-tolerant invertebrates. They characteristically travel hundreds of miles, sometimes on a daily or weekly basis, in order to exploit such invertebrate foods during the short times when the food peaks (Haig et al. 1998).

8.6 Sedimentation/ Turbidity

Sedimentation can affect birds by altering habitat structure, killing submersed vegetation, or altering the abundance or availability of prey items. (see Sections 4 and 5). However, in ponds in the interior of British Columbia, densities of breeding dabbling ducks were correlated *positively* with wetland turbidity (Savard et al. 1994).

8.7 Thermal Alteration

Thermal alteration can affect birds the greatest by preventing ice to form in some northern waters and by altering the seasonal abundance of prey items. No new studies were found documenting the direct effects of thermal alteration of wetlands on birds, but habitat changes that will occur as a result of global warming are a significant long-term concern (Poiani & Johnson 1992, Poiani et al. 1996).

8.8 Dehydration/Inundation

Dehydration/Drought

Drainage and some types of hydrological manipulation of wetlands have been well-documented as contributing causes in the decline of many wetland bird species (Batt *et al.* 1989, Bortner *et al.* 1991, David 1994, DeAngelis et al. 1997). In Manitoba, for example, wetland drainage has decreased the availability of breeding and brood-rearing areas for waterfowl (Rotella and Ratti 1992). As wetlands are drained or converted to other land cover types, local densities of wetlands decline and mean distances between individual wetlands increase. A cypress dome rehydration project in central Florida precipitated the return of 16 wetland bird species to the wetland (Weller 1995). In contrast, in the Florida Everglades, fewer bird species were found in hydrologically impacted sites than in reference sites (Gawlik and Rocque 1998).

Drought conditions also expose duck nests to greater predation as a result of diminished vegetation vigor and density, and creation of access points to islands that formerly were inaccessible (Hallock & Hallock 1993, Jobin & Picman 1997). By reducing the number and perhaps the variety of wetlands and the vegetation communities they contain, sustained regional drought or widespread draw down of water tables diminishes avian richness, bird density, and breeding success in many individual wetlands and wetland complexes (Higgins et al. 1992, Bethke & Nudds 1993, Bancroft et al. 1994, Greenwood et al. 1995, Dobkin et al. 1998). As a result of the hydrological impacts in the Everglades, wood storks begin breeding later in the season, and in most years a late start is followed by nest failures (Ogden 1994). Anthropogenic and natural reversals in wetland drying also have resulted in failed nesting attempts in wood stork colonies in South Florida (Bancroft et al. 1994)

Complexes of wetlands -- where permanently, semipermanently, seasonally, and temporarily inundated sites are close together -- are important for many waterbird species, especially in the prairie region (e.g., Rotella & Ratti 1992a,b; Cowardin et al. 1995) and in other regions where precipitation varies greatly from year to year (Meyers & Odum 1991, Fleming et al. 1994). Permanent water need not directly adjoin nesting habitat in order for larger duck species to breed successfully, and in fact sites farther from water can be more productive for Mallards, Northern Pintail, and Northern Shovelers, so long as vegetative cover is adequate (Kantrud 1993). Proximity of nest cover to permanent water may be more important to Green-winged Teal and American Wigeon. In Maine, richness of aquatic bird communities in individual wetlands was correlated with local wetland density although not with distance to the nearest wetland (Gibbs et al. 1991).

Wetlands that are inundated for only brief periods each year, and that typically lack surface connections to other water bodies (i.e., many “isolated” wetlands), are relied on almost exclusively by several bird species, especially many shorebird species. Many such areas are former wetlands that have been at least partly converted to rice fields, sod farms, soybean fields, or other agricultural activities that leave substantial areas of bare, fine-particled soil. During wet years in some regions, these wetlands also can be the only ones shallow enough to provide acceptable foraging habitat for many waterbirds, and they provide essential habitat for shorebirds most years (Nesbitt and Williams 1990, Eldridge 1992, Bishop 1992, Shuford et al. 1998, Twedt & Nelms 1999). In addition, isolated wetlands that are inundated only seasonally provide (a) vital feeding and stopover habitat for migratory waterfowl during early spring in northern regions, when many deeper wetlands remain frozen, and (b) roosting areas during high tide, when located near tidal waters (Lovvorn & Baldwin 1996, Rottenborn 1996, Burger et al. 1997). Waterbirds that use seasonal wetlands are highly mobile and adapted to exploiting "on short notice" the brief seasonal peaks of foods in these areas, provided that hydrologically diverse complexes of wetlands are maintained at a regional scale (Skagen & Knopf 1993, 1994, Farmer & Parent 1997, Oring & Reed 1997, Robinson & Warnock 1997, Warnock 1997, Haig et al. 1998, Laubhan & Gammonley 1999). Approximately 68% of the breeding bird species that utilize wetlands in the Colorado Plateau use wetlands that are inundated only briefly each year (Adamus 1993).

<u>Inundation</u>

In the south-central United States, wintering waterbird richness, abundance, and projected carrying capacity were found to be greatest on playa wetlands that were repeatedly drawn down or irrigated to a shallow depth during early April, late June, and early August, and then flooded for the winter in November. However, by initiating flooding of these playas in September, bird use occurred over a longer seasonal period (Haukos & Smith 1993, Anderson & Smith 1999). Pre-irrigation of former wetlands and fields, followed by winter flooding, was found to be effective for supporting greater numbers of wintering Northern Pintail in California's Central Valley as well (Barnum & Euliss 1991). Deeper drainwater evaporation ponds are also important to waterfowl wintering in the Central Valley (Euliss et al. 1991). Pre-irrigation tends to flush salts from underlying soils. Pre-irrigated fields, when flooded in winter, provide expansive areas where ducks are relatively safe from predation. However, in the southern Mississippi Alluvial Valley, larger concentrations of wintering shorebirds were found on permanently-inundated wetlands whose water levels were drawn down, than on fields that were flooded only during winter (Twedt et al. 1998).

Bottomland hardwood wetlands are a vital resource for many wintering birds (Zeller & Collazo 1995). Bird richness and abundance in a bottomland hardwood floodplain, both in winter and summer, were not significantly associated with relative wetness or flooding duration (Wakeley & Roberts 1994). Flooding of bottomland wetlands displaced wintering American Woodcock and might have contributed to increased mortality (Krementz *et al.* 1994). The breeding success of large wading bird species nesting near Lake Okeechobee, Florida, varied between years depending on annual drought or flood conditions. The particular response was species-specific (Smith and Collopy 1995). Prolonged hydroperiods may be necessary for the development of populations of large fish (Fleming et al. 1994) that are selected for by Wood Storks, Great Egrets, and Great Blue Herons. Avian richness in restored prairie wetlands was mostly greater during a wet than a dry year (Hemesath & Dinsmore 1993).

Wetlands created by beaver contain significantly more waterbird species than inactive beaver sites or potential beaver sites of the same size (Medin and Clary 1990, Grover and Baldassarre 1995). Beaver affect the species richness by impounding water and creating a diverse mosaic of emergent vegetation, flooded dead trees, and open water. Beaver ponds comprising only 25% of the wetland area in Ontario were used disproportionately by dabbling ducks (Merendino *et al.* 1995). In an examination of 70 beaver ponds, Grover and Baldassare (1995) determined that active beaver ponds had more species of birds than inactive sites. Active sites had more open water, dead standing trees, surface water, and flooded emergents than inactive sites. Female mallards in the St. Lawrence River Valley spent most of their time breeding in forested-live wetlands (40%) and postbreeding time in forested-dead wetlands (35%) (Losito et al. 1995).

The diking and filling above the average high water level of coastal wetlands on the Great Lakes has rendered much former habitat almost useless for nesting waterbirds (Prince *et al.* 1992). Construction of reservoirs also impacts birds by eliminating many wetlands, while creating others with less stable water levels (Nilsson & Dynesius 1994). Associated changes in river morphology influence species composition of wintering waterfowl (Johnson et al. 1996). In the Florida Everglades, reproductive success of an endangered subspecies -- Cape Sable Seaside Sparrow -- was less during a period when water levels were kept at unusually high levels, potentially flooding nests, increasing nest predation, and changing vegetation composition and structure (Nott et al. 1998).

8.9 Effects of Vegetation Removal, Habitat Fragmentation

Much ornithological research during the past decade has focused on impacts of loss of natural vegetative cover on songbird reproductive success. Most of these studies have focused on upland forests, but an increasing number have examined grasslands, riparian systems, and floodplains. In any event, most findings from upland landscapes are probably transferable to wetlands with similar vegetation structure. Many studies of upland forest tracts of various sizes (areas) situated in agricultural or urban landscapes continue to document a striking decrease (in the smaller, partly fragmented tracts) in the occurrence, abundance, and/or reproductive success of several neotropical migrant songbird species (e.g., Andren 1994, Askins 1995, Donovan et al. 1995, 1997, Friesen et al. 1995, McIntyre 1995, Fauth et al. 2000). This is widely attributed to disproportionate vulnerability of these species to predation and parasitism in smaller tracts (Robinson et al. 1995), although some evidence from the floodplain forests of the Upper Mississippi River suggests that predation on nests may actually be less in smaller isolated patches (Knutson et al. 2000). Wooded patches smaller than about 100 hectares, and especially those smaller than 16 hectares, generally do not support the full set of songbird species present in larger wooded patches (Blake & Karr 1984, Robbins et al. 1989). Some breeding species return annually to their natal sites even after their former habitat has been severely altered (Villard et al. 1995). Use of bird species assemblages as indicators of vegetation disturbance in non-wetland environments is discussed by Hutto (1998).

From several studies it is now evident that what has been documented in uplands -- that larger connected patches of natural habitat, relatively unfragmented by roads, support more native bird species than smaller, fragmented patches -- is also true in wetland and riparian areas. These wetland studies include: (a) a survey of 30 Ontario wetlands ranging in size from 13 to 1500 hectares (Findlay & Houlahan 1997), (b) an analysis of data from 18 forested wetlands in Maryland (Schroeder 1996), (c) a survey of wetland and riparian habitat on 158 New England lakes (Allen & O'Connor), (d) a survey of 40 Minnesota wetlands (Whited et al. 2000), and (e) a survey of urban riparian areas in central California (Rottenborn 1999). Riparian areas and wooded wetlands -- the habitats that often are often most-used as breeding habitat by neotropical migrant birds (Gates and Giffen 1991, Flather & Sauer 1996) -- also happen to be characterized by an unusually large variety and density of nest predators and parasites. Neotropical migrant birds include dozens of species of warblers, vireos, tanagers, and other long-distance migrants.

As a group these comprise a large portion of the breeding avifauna in most regions, and are declining throughout developing regions in North America.

In addition to being sensitive to variation in land cover at a landscape scale, bird species composition and density are very sensitive to structure and age of vegetation within a particular site (e.g., Hanowski & Niemi 1990, Craig & Barclay 1992, Edwards & Otis 1999, Hanowski et al. 1999). Bird richness and community structure have been compared among wetlands having different vegetation cover types (e.g., Gibbs et al. 1991, Craig & Barclay 1992, Adamus 1992), but comparisons can be confounded by differences among cover types with regard to bird species detectability (Morrison et al. 1998). In Florida, riparian forests along blackwater rivers had a somewhat different species composition than riparian forests along spring-fed rivers. Spring-fed sites had more species, possibly because of their more complex vertical structure and greater canopy closure (Leonard 1994). Neotropical migrants are especially sensitivity to the structural complexity of habitat in bottomland hardwood forests (Pashley & Barrow 1993). Bird species also differed among Colorado riparian areas with different vegetative structure (Finch 1991) and in New Mexico riparian stands of different ages (Farley et al. 1994). Vegetation structure clearly influenced species composition in central Iowa, where 48 breeding species were observed along grassed waterways, but only 14 in surrounding crop fields. Breeding bird densities in the 44 grassed waterways were more than 3 times greater than in crop fields. Bird use peaked in July, suggesting that grassed waterways should not be mowed until the end of August or early September (Bryan and Best 1991).

Vegetation composition of islands located in lakes and wetlands also determines bird species that nest there. Low shrubs, tall grasses, and dense herbaceous plants provide nesting cover for Canada Goose, Mallard, Gadwall, and other ducks. Mallard, Blue-winged Teals, and Gadwall in a North Dakota marsh had the highest nesting density and success on islands that were 335-1,085 m offshore, surrounded by 150-200 m of open water, and contained tall, dense, brush or forb cover (Williams and Crawford 1989). Other species that prefer dense vegetation in wetlands include Northern Harrier, Short-eared Owl, and Ring-necked Pheasant (Homan et al. 1993). One North Dakota study that used herbicides to reduce vegetation cover found a reduction in densities of Marsh Wren, Red-winged Blackbird, and Common Yellowthroat, up to two years after application (Linz et al. 1993, Blixt et al. 1993). A Minnesota study found no positive correlation between cover ratio (the ratio of open water to emergent vegetation) and numbers of Yellow-headed Blackbird, Song Sparrow, or Sora (Olson 1992). However, avian richness in prairie wetlands cannot always be predicted by vegetation structural diversity (Olson 1992).

The conversion of forested wetlands to emergent and open water wetlands can alter species composition and richness of breeding birds both on-site and locally. For example, 53% of the bird species that formerly used forested wetlands no longer occur regularly where such forests have been mined and converted to emergent wetlands (Doherty 2000). A sustainable supply of standing dead trees (snags) also is important in forested wetlands (Sedgwick & Knopf 1990).

Vegetated wetlands in Ontario were used to a greater degree than open water areas by Mallard, American Black Duck, Wood Duck, Blue-winged Teal, and many other dabbling duck species (Merendino *et al.* 1993, Merendino and Ankney 1994). Similarly, Connecticut tidal marshes with a larger ratio of vegetation to open water contained more breeding bird species, as expected (Craig & Beal 1992). However, most wading bird species avoid dense stands of vegetation (Bancroft et al. 1994, Hoffman et al. 1994, Smith et al. 1995). Across a size range of 0.01 to 1.3

hectares, shallow ponds created in tidal salt marshes for mosquito control were more attractive to migrating waterfowl when they were smaller than 0.02 hectares or larger than 0.08 hectares (Erwin et al. 1991). Migrating shorebirds were most attracted to ponds larger than 0.10 hectares

In most instances, achievement of successful wetland restoration is clearly tied to the recovery of vegetation structure (Brown & Smith 1998, Zedler 1993, 1998) and the maintenance of complexes of suitable wetlands in an appropriate landscape context (Morrison et al. 1994). Thus, birds have sometimes been used to monitor progress in the condition of wetlands following restoration (e.g., Sewell and Higgins 1991, Weller 1995). Breeding bird communities of natural prairie potholes have been found to be more diverse than those of recently restored wetlands (Delphey & Dinsmore 1993, VanRees-Siewert & Dinsmore 1996), although numbers specifically of breeding ducks may not differ (Delphey & Dinsmore 1993). The lack of well-developed vegetation zones that typify natural potholes likely led to the lower numbers or less frequent encounters of American Goldfinch,Virginia Rail, Sora, Least Bittern, American Bittern, Common Yellowthroat, Swamp Sparrow, and Red-winged Blackbird in recently restored wetlands (Delphey and Dinsmore 1993). The development of a diverse community of submersed aquatic plants may contribute to recovery of avian communities in some restored and constructed wetlands (Weller et al. 1991, Leschisin et al. 1992, Mulyani & DuBowy 1993). In peat-mined Canadian bogs that remained less vegetated than unmined bogs even 20 years after abandonment, at least 10 species were less common in the bogs that had been mined, especially using vacuum techniques (Desrochers et al. 1998).

Even when in pristine condition, wetlands are not necessarily covered completely with vegetation at all times, and very dense stands of vegetation are unsuitable for several species (Olson 1992, McMurl et al. 1993, Hemesath & Dinsmore 1993, Blixt et al. 1993). For example, migrating shorebirds, as well as American Robin and Grasshopper Sparrow, were observed in newly restored wetlands more frequently than in natural wetlands, probably due to their aversion of dense vegetation (Delphey & Dinsmore 1993). As dense stands of vegetation are thinned, the diversity of bird species using a wetland typically increases or remains stable (Blixt et al. 1993), especially if open water begins to occupy spaces cleared in the vegetation. Small floating mats of dead herbaceous vegetation, interspersed with open water, are important to some waterbirds (Linz et al. 1997). In the Dakotas and Montana, islands in wetlands also had greater nest densities and greater nest success than surrounding uplands (Lokemoen and Woodward 1992).

Logically, the greatest differences between bird composition in riparian/wetland habitats vs. adjoining upland habitats have been found where the adjoining habitats are most dissimilar in terms of vegetation structure (Strong & Bock 1990, Gates and Griffen 1991, Hooper 1991, McGarigal & McComb 1992, Murray & Stauffer 1995, Karriker 1996). This is true in agricultural as well as wooded landscapes. For example, in Saskatchewan wetlands, more birds were found in natural wetlands and wetlands surrounded by organic farms, than in wetlands surrounded by conventional farms or in wetlands surrounded by farms that minimized tillage but still used chemicals (Shutler et al. 2000).

Even when vegetative structure of wooded sites does not change between years, species composition within the sites may change if land cover in the surrounding landscape is altered (Triquet et al. 1990, Richter & Azous 2001). Forested South Carolina wetlands surrounded by pine woods supported several area-sensitive breeding species at the expense of species that prefer field edges (Kilgo et al. 1997). Forested riparian areas surrounded by intact forests had somewhat different bird species than those that were not, in Kentucky (Triquet et al. 1990), Pennsylvania (Croonquist & Brooks 1993), Vermont (Meiklejohn & Hughes 1999), and Idaho (Saab 1999). Riparian strips and wooded wetlands have a more unique avifauna in regions with homogeneous land cover, whether the homogeneity is represented by forests nearly unbroken by logging and roads (LaRue et al. 1995, Anthony et al. 1996) or by agricultural land unbroken by hedgerows and other patches of natural vegetation. Nonetheless, riparian strips usually do not provide the same quality habitat or support the same richness and abundance of species as large, wide tracts of lowland forest (Johnson & Brown 1990, Whitaker & Montevecchi 1999).

Several studies have attempted to identify (a) types and configuarations of edges between habitats that minimize predation of bird nests, (b) minimal and optimal widths of riparian cover necessary to maintain songbird richness at multiple scales, and (c) indicator species or bird community metrics (response variables) that statistically are most sensitive to particular types of human influences, at particular scales. In bottomland hardwood forested wetlands, predation on bird nests is greater at forest-cropland edges than at forest-river edges or at the edge between forest and natural levees (Saracco & Collazo 1999). Studies of buffer width basically have concluded "the wider the better" if the aim is to protect the largest possible component of a region's avifauna. However, in agricultural landscapes, the number of individual breeding birds *per unit area* in wider corridors is not necessarily greater -- only the number of species (Rich et al. 1994, Darveau et al. 1995, Thurmond et al. 1995, Kilgo et al. 1998, Meiklejohn & Hughes 1999, Whitaker & Montevecchi 1999). The following table summarizes recent North American research on bird response to riparian and wetland buffer widths.

Buffer or corridor width (includes both stream banks)	Function	Habitat	Location	Researcher (s)
> 60 m	to sustain forest-dwelling birds	coniferous riparian forest (balsam fir)	Quebec	Darveau *et al.* (1995)
20 m	to support several ubiquitous species but not most forest-dwelling species	coniferous riparian forest (balsam fir)	Quebec	Darveau *et al.* (1995)
411 ft	to support full complement of species	riparian & wetland	Pennsylvania	Croonquist and Brooks 1993
164 ft	to support many sensitive species	riparian & wetland	Pennsylvania	Croonquist & Brooks (1993)
14 ft	to maintain portions of the bird community in disturbed areas	riparian & wetland	Pennsylvania	Croonquist & Brooks (1993)
40 m	to maintain canopy-sensitive species in forested headwater areas	coniferous & mixed	Oregon	Hagar (1999)
>500 m	to maintain the complete avian community characteristic of South Carolina bottomland forests	hardwoods		Kilgo et al. (1998)
> 100 m	to maintain a probability of occurrence of at least 50% of most breeding species	hardwoods	Maryland	Keller et al. (1993)
>100 m	to maintain breeding neotropical migrant species	hardwoods	Georgia (Altamaha River)	Hodges & Krementz (1996)
175 ft	did not support forest-interior neotropical migrant	hardwoods		Thurmond et al. (1995)

	species			
30-50 m	to maintain bird richness at a landscape scale	hardwoods	eastern Texas	Dickson et al. (1995)
75-175 m	to include 90% of species breeding in a region's riparian habitats	mixed	Vermont	Spackman & Hughes (1995)
> 60 m	to maintain wintering populations of Hooded Warbler and Acadian Flycatcher	urban riparian	Florida	Leonard (1994)

On a related note, a few studies have examined the apparent reluctance of forest birds to cross areas of non-forest habitat, i.e., "gaps" that lack corridors (Machtans et al. 1996). During the post-fledgling period, several songbird species in Quebec were only half as likely to cross a 50-m wide field as cross a 50-m wide wooded area (Desrochers & Hannon 1997). Wooded routes were preferred even when 3 times longer than shortcuts through openlands. Gaps narrower than 30 m did not function as barriers to movements of most songbirds. During the breeding period, species that characteristically breed in forest interiors were least likely to cross gaps wider than 25 m (Rail et al. 1997). Similar results were found in the same region among wintering songbirds (St. Clair et al. 1998). In Georgia, at least 5 breeding species avoided crossing the 100-m wide Altamaha River (Hodges & Krementz 1996). In North Dakota, post-fledgling birds were found to move between wooded shelterbelts more often when the shelterbelts were connected by wooded corridors than when structurally similar shelterbelts with the same separation distance and area were not connected with corridors (Haas 1995). After clearcutting of an Alberta forest, juvenile birds moved along the remaining riparian corridors (Machtans et al. 1996). Connectivity of natural cover at a landscape scale was a statistically important predictor of bird species composition amid agricultural regions of Minnesota (Whited et al. 2000).

In contrast, in urban and desert landscapes of Arizona, where many bird species breed almost exclusively in riparian areas (Germaine et al. 1998), the importance to migrant birds of wooded connectivity (i.e., lack of gaps) was found to be less than the influence of total wooded acreage, regardless of whether the wooded acreage is situated along a stream or as an oasis (Skagen et al. 1998). Although migrating songbirds used riparian habitats extensively, they did not appear to be using riparian corridors as travel lanes during migration. Also, in more temperate forested landscapes of the western United States, no clear evidence has been found linking forest fragmentation to reduced avian abundance or productivity (Schieck et al. 1995, Tewksbury et al. 1998).

Effects of habitat fragmentation on songbirds are not limited to wooded habitats. In grassland landscapes of Nebraska, herbaceous wetlands that are narrow (e.g., perimeter-area ratio <0.01) support fewer breeding species, as do herbaceous wetlands smaller than about 30 hectares (Helzer and Jelinski 1999). Narrow wetlands tended to have fewer individuals of Red-winged Blackbird, Dickcissal, Upland Sandpiper, Western Meadowlark, Grasshopper Sparrow, and Bobolink, and the latter 4 species also were less likely to use small wetlands. Areas more than 1 km from roads in Massachusetts were avoided by grassland birds (Forman et al. 2000). Wetland area also was a significant predictor of breeding bird richness in herbaceous tidal marshes of Connecticut (Craig & Beal 1992) and inland wetlands of Maine (Gibbs et al. 1991). Nest success of some Missouri species also was found to be less in smaller grassland patches (Winter & Faaborg 1999). In prairie pothole wetlands of eastern South Dakota, species that are most vulnerable to loss of small wetlands are very mobile species that exploit resources over a broad region (Naugle et al. 2000). The excessive removal of vegetation by livestock -- regardless of the size, connectivity, and shape of wetland in which it occurs -- can reduce onsite avian diversity and reproductive success of some species (Anderson 1993, Ohmart 1994, Gilbert et al. 1996, Weller 1996).

Predation of bird nests in grassland and other open habitats has been shown to be greater (a) close to edges with woods or plowed fields (Johnson & Temple 1990, Burger et al. 1994, Pasitschniak-Arts & Messier 1995, 1996, Bollinger and Peak 1995), (b) in smaller patches of natural grassland than in large patches (Kantrud 1993, Burger et al. 1994), (c) where human trails go through sparse cover (Olson & Rohwer 1998, Miller et al. 1998), (d) in grazed areas (Gilbert et al. 1996), and (e) where dikes or trails built on fill within a wetland make it easier for predators to gain access (Peterson & Cooper 1991). Predation also may be less in large wetlands because they are more likely to contain stretches of water too wide and deep for some mammalian predators to cross (Picman et al. 1993, Picman & Schriml 1994, Esler & Grand 1993). Documented high rates of nest parasitism and predation are believed to contribute significantly to a long-term decline of Song Sparrow in wetlands of coastal British Columbia and possibly elsewhere (Smith & Arcese 1996, Rogers et al. 1997, Larison et al. 1998), as well as to a decline of Willow Flycatcher in some western riparian areas (Harris 1991, Brown 1994). Suburban ditches and hayfields near Boulder, Colorado, were repeatedly a sink rather than a source of nesting Red-winged Blackbirds, due mainly to nest predation rather than cowbird parasitism (Vierling 2000). In contrast, natural habitats (wetlands and tallgrass prairie) with fewer nearby buildings were a source. Parasitism of Red-winged Blackbird nests by Brown-headed Cowbird in Iowa was less at restored wetlands probably because of the lack of mature trees from which to perch and search for nests (Delphey & Dinsmore 1993).

In southeastern regions where bottomland hardwood forests were once common, their removal and fragmentation at a large scale has been associated with local or regional reduction in abundance of the following breeding species (Burdick et al. 1989, Smith & Schaefer 1992, Thurmond et al. 1995, Kilgo et al. 1998, Dickson et al. 1995):

> Mississippi and Swallow-tailed Kites; Red-shouldered Hawk, Barred Owl, Ruby-throated Hummingbird; Pileated, Red-bellied, and Downy Woodpecker; White-breasted Nuthatch, Great-crested Flycatcher, Yellow-throated Vireo, Prothonotary Warbler, Northern Parula, Swainson's Warbler, American Redstart, Black-and-White Warbler, Summer Tanager

Species that prefer thickets, such as White-eyed Vireo, Hooded Warbler, and Northern Cardinal, may increase after logging. Species that are more prevalent in young stands (20-30 years old) -- including Acadian Flycatcher, Red-eyed Vireo, and Blue-gray Gnatcatcher (Wigley and Roberts 1994, Mitchell 1989, Kilgo et al. 1998) – also seem to tolerate narrower riparian strips. Wood Thrush and Louisiana Waterthrush often occur in narrow wooded riparian areas in South Carolina (Kilgo et al. 1998). However, in eastern Texas landscapes and perhaps elsewhere, some of these species have been found exclusively or mostly in the widest riparian zones (Dickson et al. 1995).

In northern regions, breeding species that appear to be less common in narrow than in wide riparian strips or woodlots include the following (from Johnson & Brown 1990, Darveau *et al.* 1995, Spackman & Hughes 1995, Meiklejohn & Hughes 1999, Whitaker & Montevecchi 1999):

Pileated Woodpecker, Yellow-bellied Flycatcher, Red-breasted Nuthatch, Veery; Swainson's and Hermit Thrushes, Golden-crowned Kinglet, Ovenbird; and Blackpoll, Black-throated Green, Bay-breasted, Blackburnian, and Cape May Warbler

Severe grazing, mowing, fire, or herbicide application at inappropriate times is detrimental to many waterbird species and recovery of bird diversity in grazed areas may take several years (Schultz & Leininger 1991, Johnson et al. 1991, Higgins et al. 1992, Gilbert et al. 1996, Dobkin et al. 1998, Warkentin & Reed 1999). However, some studies (Clary & Medin 1993, Medin & Clary 1990, 1991) found no statistically significant decline in overall avian richness and/or abundance associated with grazing, or found reduced nest density but not reduced nest success (Bowen & Kruse 1993). Effects depend on the grazing regime, the wetland plant community that is being grazed, and other factors (Barker et al. 1990, Sanders & Edge 1998). In western rangelands, wetland and riparian species that appear to be most sensitive to the immediate effects of grazing include: Wilson’s Phalarope, Willow Flycatcher, Yellow-breasted Chat, Yellow Warbler, Song Sparrow, Savannah Sparrow, Spotted Towhee, and Red-winged Blackbird (Bock et al. 1993, Dobkin et al. 1998, Sanders & Edge 1998). Even the reduction of vegetative cover by native deer can adversely affect nesting songbirds; deer densities of 8-15 deer per km^2 (about 20-40 per mi^2) were found to be the least at which widespread effects on songbird richness and abundance were noted in Pennsylvania (DeCalesta 1994).

8.10 Disturbance from Human Visitation and Other Influences

Frequent visitation of wetlands by boaters and other recreationists can adversely some waterbirds, especially near nesting colonies (Dahlgren & Korschgen 1992, Erwin et al. 1993, Klein 1993, Knight & Gutzwiller 1995, Klein et al. 1995, Rogers & Smith 1997). Human intrusion can disrupt bird feeding patterns (Skagen et al. 1991), reduce bird song which is vital to reproductive success of most species (Gutzwiller et al. 1994), and cause at least temporary shifts in bird community richness and abundance (Riffell et al. 1996). Powerlines and transmission towers are also a source of mortality (Bevanger 1998).

Sport fishing has been suggested as a possible cause of a shift in the fish population from larger predatory fish to smaller fish in a Canadian lake, and the resultant shift in avian species composition (Gerrard *et al.* 1993). There were associated large increases in the abundance of Common Loon, American White Pelican, Osprey, Great Blue Heron, and Herring, Ring-billed, and Bonaparte's Gulls. No changes were seen in the abundance of mergansers, Bald Eagle, or Common Tern.

Introduction of non-native fish, invertebrates, and plants also can affect wetland birds. For example, large herbivorous fish such as carp compete directly with birds for submersed aquatic plants, and also reduce water clarity (Bouffard & Hanson 1997). The introduced Zebra Mussel provides abundant food for some diving duck species, but indirect effects on food chains are less well known (Custer & Custer 1996, Hamilton & Ankney 1994). The introduced waterweed, *Hydrilla,* also has apparently benefitted some species but its effects on others are unknown (Esler 1990). Although purple loosestrife (*Lythrum salicaria*) has commonly been considered to have few wildlife values, American Goldfinches nest in this habitat in New York wetlands (Kiviat 1996).

Global climate change has enormous potential to influence wetland birds, by influencing the stressors described in the rest of this section. Potential effects are discussed by Larson (1993).

8.11 Wetland Monitoring

Spatial and Temporal Variation

Choice of appropriate sample sizes depends on measured variation in the target taxa and metrics. Such coefficients of variation were calculated from previous avian studies in prairie pothole wetlands (Adamus 1996), and are summarized from various studies elsewhere at: http://www.im.nbs.gov/powcase/powvariation.html

In a central Maine bog, breeding birds appeared to select slightly different microhabitats during consecutive years (Wilson et al. 1998). The same was noted for some wintering waterfowl species in southern forested wetlands (Kaminski et al. 1993) and for avian richness in some western riparian habitats (Sanders & Edge 1998). Variation in use of 87 Maine wetlands during an 8-year period by 15 waterbird species was quantified by Gibbs et al. (1991). Interannual variation in avian species richness in North American wetlands of various types surveyed by the Breeding Bird Census program was compiled by Adamus & Brandt (1990). Common pitfalls in the statistical analysis of multi-year data on animal use of habitats are described by Schooley (1994). Also, in 6 Connecticut wetlands, 12 species changed their habitat associations among seasons (Craig & Barclay 1992).

Changes in the avifauna of a series of prairie pothole wetlands on the scale of decades were determined by Igl and Johnson (1997). Detecting a 5-10% decline in populations of some shorebird species would require more than 10 years of survey effort, due to the difficulty of surveying the highly mobile populations (Warnock et al. 1998). The importance of surveying migratory shorebirds at very large spatial scales (e.g., regions) is highlighted by Haig et al. (1998). Perhaps the most spatially-extensive wetland bird survey was conducted by Naugle et al. (2000), who characterized 834 prairie wetlands and recommended region-scale wetland conservation priorities based on habitat relationship models derived from statistical analysis of these data.

Techniques

Some information applicable to surveying wetland birds is presented in Bibby et al. (1992), Ralph et al. (1993, 1995), Gibbs & Melvin (1993), Ribic et al. (1999), Weller (1999), and Adamus (2001). Whenever possible, surveys conducted for comparing wetlands or wetland types using waterbirds should be done at night as well as during daytime, because habitat selection often differs dramatically (Beyer & Haufler 1994, McNeil et al. 1992, Anderson & Smith 1999). Technological advances such as radiotelemetry, remote time-lapse tape recording and photography, and molecular markers are increasingly being applied not only to count birds, but also to determine their movements and quantify their persistence and breeding success in diverse wetlands, e.g., whether particular wetlands are “sources” or “sinks” for local birds. Where birds that nest in cavities are present, bird boxes provide a convenient means of monitoring reproductive success, with minimal disturbance and without the labor of having to find nests. They have been used successfully to monitor impacts from heavy metals (Kraus 1989, Peterson and McEwan 1990) and acid precipitation (St. Louis and Barlow 1993). Another approach -- time-budget analysis -- involves documenting the hours a wetland is visited by various species and usually requires purchase and installation of video equipment that automatically photographs portions of the wetland at specified intervals. From viewing the tapes, the duration of each activity (e.g. feeding) of visible birds in each photographed zone can be determined. It is costly to implement for studies intended to survey more than a few wetlands. However, time-budget analysis has demonstrated that estimates of bird density are not necessarily sufficient to indicate a degraded wetland condition, i.e., a wetland with diminished invertebrate densities (Eldridge & Krapu 1993). Even more labor-intensive are studies that employ banding (Meyers & Pardieck 1993). Over long periods of time, they can provide information on population dynamics of particular species, enabling judgments of whether a wetland is a “source” or “sink” for breeders.

Metrics for Assessing Impacts to Wetland Bird Communities

Information on this topic is reviewed on EPA’s wetland biomonitoring web page (Adamus 2001).

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